1974-11-01

Management Programs, Research and Effects of Present Land Use Activities on Water Quality of the Great Lakes: Volume 1 of 2 Volumes

International Reference Group on Great Lakes Pollution From Land Use Activities

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INTERNATIONAL REFERENCE GROUP ON GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES

INTERNATIONAL JOINT COMMISSION

MANAGEMENT PROGRAMS, RESEARCH AND EFFECTS OF PRESENT LAND USE ACTIVITIES ON WATER QUALITY OF THE GREAT LAKES NOVEMBER 1974 — VOLUME I

INTERNATIONAL REFERENCE GROUP ON GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES

MANAGEMENT PROGRAMS, RESEARCH AND EFFECTS OF PRESENT LAND USE ACTIVITIES ON WATER QUALITY OF THE GREAT LAKES

VOLUME 1 of 2 VOLUMES

1/2. Residential, Commercial and Industrial Areas
3. Transportation
4. Extractive Areas
5. Pesticides and Herbicides
6. Nutrients
7. Erosion and Sedimentation
8/9. Animal Wastes and Intensive Animal Feedlots

NOVEMBER, 1974

Reprinted August, 1976

Prepared by the U.S. Section of Task Group A for the
POLLUTION FROM LAND USE ACTIVITIES REFERENCE GROUP OF THE
INTERNATIONAL JOINT COMMISSION

PREFACE

The Great Lakes Water Quality Agreement, with Annexes and Texts and Terms of Reference Between the United States of America and Canada, signed at Ottawa on April 15, 1972, included a reference to study pollution in the Great Lakes System from agricultural, forestry, and other land use activities. The reference asked that the study assess whether the boundary waters of the Great Lakes System were being polluted by land drainage and if so, what remedial measures would provide improvements in controlling pollutants from land usage. The need for better definition of the impact of land use activities, practices, and programs on water quality in the Great Lakes area had become increasingly magnified. Through the Agreement, both the United States and Canadian governments requested the International Joint Commission to investigate land use activity impacts upon the Great Lakes. Accordingly, the International Reference Group on Great Lakes Pollution From Land Use Activities was established in February, 1973, and produced a detailed study plan (February, 1974) outlining an intensive study, scheduled for completion by 1977.

Detailed Study Plan, February, 1974

The study plan emphasizes four main tasks:

Task A: To assess problems, management of programs and research and to attempt to set priorities in relation to the best information now available on the effects of land use activities on water quality in boundary waters of the Great Lakes.

Task B: Inventory of land use and land use practices, with emphasis on certain trends and projections to 1980 and, if possible, to 2020.

Task C: Intensive studies of a small number of representative watersheds, selected and conducted to permit some extrapolation of data to the entire Great Lakes basin and to relate contamination to water quality, which may be found at river mouths on the Great Lakes, to specific land uses and practices.

Task D: Diagnosis of degree of impairment of water quality in the Great Lakes, including assessment of concentrations of contaminants of concern in sediment, fish and other aquatic resources.

U. S. Task A Study Plan

In order that subsequent tasks may derive full benefit of past and present programs and activities pertinent to the goals of the Reference Group, it was felt to be essential to assess the current state-of-the art or knowledge about land drainage and to inform the Reference Group of current developments. The objectives of Task A were to analyze the pollution problem and potential of various land uses and to document the practicality of alternative, remedial, or control measures.

In order to meet these objectives for the U. S. portion, studies were agreed upon by Task A members to provide a state-of-the-art assessment for the following categories:

A1: Residential areas
A2: Commercial and industrial areas
A3: Transportation
A4: Extractive areas
A5: Pesticides and herbicides
A6: Nutrients
A7: Erosion and sedimentation
A8: Animal wastes
A9: Intensive animal feedlots
A10: Forestry
A11: Recreation land
A12: Undeveloped land
A13: Liquid waste disposal
A14: Solid waste disposal
A15: Land Fills (dredging activities)
A16: Deepwell disposal
A17: Management and control of land use/water quality problems

Selection of Principal Investigators

Limited amount of funding available from the U. S. Environmental Protection Agency, lead agency for the U. S. Task A effort, required some support from other Federal agencies. The U. S. Department of Agriculture cooperated by having the Soil Conservation Service support Category A7 - Erosion and Sedimentation, and the Forest Service Category A 10 - Forestry. For the other categories, the GLBC staff developed a listing of consulting firms, Universities, and individuals knowledgeable and interested in the various study areas. Requests for proposals at a limited fixed-price were initiated and invitations sent out to all competent to deliver the professional services. After submittal of proposals and professional capabilities, the Task A Group selected the potential principal investigators and associates. The Great Lakes Basin Commission, under contract with the U. S. Environmental Protection Agency, negotiated subcontracts with the organizations and principal investigators. The last category, A17 - Management and control of land use/water quality problems with focus on the Great Lakes States programs, was developed by the Great Lakes Basin Commission staff.

U. S. Task A Study Participants

The lead agency for Task A on the U. S. portion has been the U. S. Environmental Protection Agency with the participation of the Great Lakes Basin Commission, and the U. S. Department of Agriculture's Soil Conservation Service and Forest Service. The Great Lakes Basin Commission acted as the main contractor for fifteen of the categories, while the Department of Agriculture contributed two categories to the U. S. effort.

U.S. members of the Land Use Activities Reference Task A Group included:

Dr. Richard R. Parizek, Pennsylvania State University
Mr. James P. Dooley, Michigan Department of Natural Resources
Mr. John Pegors, Minnesota Pollution Control Agency
Mr. Eugene A. Jarecki, GLBC Technical Representative
Mr. Fred O. Sullivan, U.S. EPA Project Officer

The principal investigators of the U.S. Task A technical papers were:

Task A1-A2: Dr. John E. Schenk, Environmental Control Technology Corporation, Ann Arbor, Michigan
Task A3: Dr. Robert C. Stiefel, Ohio State University, Columbus, Ohio
Task A4: Dr. Robert L. Bates, Ohio State University, Columbus, Ohio
Task A5: Dr. Gordon Chesters, University of Wisconsin, Madison, Wisconsin
Task A6: Dr. D.E. Armstrong, University of Wisconsin, Madison, Wisconsin
Task A7: Mr. William Mildner, USDA, Soil Conservation Service, Hyattsville, Maryland
Task A8-A9: Dr. Raymond C. Loehr, Cornell University, Ithaca New York
Task A10: Mr. Robert Lindahl, USDA, Forest Service, St. Paul, Minnesota
Task A11: Dr. John Harkin, University of Wisconsin, Madison, Wisconsin
Task A12: Dr. Charles E. Herdenorf, Ohio State University, Columbus, Ohio
Task A13: Dr. Robert H. Miller, Ohio State University, Columbus, Ohio
Task A14: Dr. Charles A. Moore, Ohio State University, Columbus, Ohio
Task A15: Dr. Charles A. Moore, Ohio State University, Columbus, Ohio
Task A16: Dr. Wayne A. Pettyjohn, Ohio State University, Columbus, Ohio
Task A17: Mr. Robert W. Reed, Great Lakes Basin Commission, Ann Arbor, Michigan

Assistance in development and review of various phases of Task A is acknowledged by Mr. Merle Tellekson, Mr. Gene Pinkstaff, and Mr. John McGuire from the U.S. Environmental Protection Agency, Region V, Mr. Gerald B. Welsh, USDA-Soil Conservation Service, and Mr. Robert W. Reed of the Great Lakes Basin Commission staff. The Canadians who were particularly helpful include Mr. Kim Shikaze, Environment Canada, Ottawa, and Mr. Martin G. Wood from the Ontario Ministry of the Environment, Toronto. Considerable assistance was also provided by Mr. Pat Chamut of the International Joint Commission Regional Office in Windsor, Ontario.

Each report has been reviewed by the Task A Group and its comments considered before approval for final report development and submittal to the U.S. Environmental Protection Agency for meeting contractual terms. Some

reports were reviewed by the International Reference Group on Great Lakes Pollution From Land Use Activities. These reports form the U. S. contribution to the Task A effort. The findings and conclusions of each report reflect the principal investigators' findings and do not necessarily reflect the opinions or positions of their respective agencies, those of the Reference Group, or those of the U.S. Environmental Protection Agency.

NOTE: A joint summary report of the Canadian and United States Task A efforts presenting the findings is being prepared.

Land Use/Water Quality Relationships in the U.S. Great Lakes Basin
Task A: To assess problems, management of programs and research...

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Combined Categories: A1 - Residential Areas
A2 - Commercial and Industrial Areas

THE EFFECT OF RESIDENTIAL AND COMMERCIAL-INDUSTRIAL LAND USAGE ON WATER QUALITY

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for the
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Ann Arbor, Michigan

To be used as portion of the U. S. Task A Report on GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by the International Joint Commission - prepared in partial fulfillment of U. S. Environmental Protection Agency Contract No. 68-01-1598

August 1974

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Section 1

SUMMARY

1.1 Introduction

Residential and commercial-industrial land usage (collectively referred to as urban land usage) has long been recognized qualitatively as a major factor in the water quality of the Great Lakes. In general, the effect on water quality of residential land usage will not be the same as commercial-industrial usage. Due to their general intermixing geographically, however, it is difficult to separate the effects of each individual usage, and it is thus convenient to consider them together.

This type of land usage is obviously scattered throughout the Great Lakes Basin, however it is much more highly concentrated around Lakes Michigan, Erie, and Ontario, and the Saginaw Bay portion of Lake Huron.

The unending cycle of water in the natural environment is characterized by contamination to purification and back to contamination. Precipitation through the atmosphere results in a cleansing of the air by dissolving contaminants and physically washing out particulate material[1, 2]. Overland flow of runoff to water courses adds further soluble and insoluble naturally occurring materials to the water[2,3]. Percolation of water into the soil and subsurface flow also can add pollutional material[2]. Thus, even without the affect of human activity through the development of residential and commercial-industrial areas, pollution of land drainage occurs.

The urbanization resulting from residential and commercial-industrial development magnified the pollution potential of drainage both qualitatively and quantitatively. This occurs due to the larger amounts of pollution-inducing materials resulting from this type of activity, as well as enhancing the rate of runoff which provides greater scour potential

and magnifies the shock loading on receiving streams. In addition, the "people-generated" pollution in these areas will often contain substances not normally released in nature. The frequent transport of storm drainage and municipal waste in the same conduit also allows for the discharge of highly pollutional raw wastewater during periods of extreme runoff (4) Those materials resulting from this urban and semi-urban land usage which affect water quality may be separated into a number of categories, as shown in Table I.

The degree to which pollution from urban drainage affects the Great Lakes and their connecting waters is dependent upon a number of factors. One of the chief factors is the location of the urban activity in relation to the waters of the Great Lakes. Drainage which enters a tributary to the Great Lakes will be considerably modified during its residence in the tributary. If the residence term prior to discharge to the lakes themselves is sufficiently long, the bacterial pollution and possibly the organic pollution (as measured by BOD) will have been at least partially exerted. Similarly, the heavier solids will tend to be deposited in the tributary and thus removed from the water mass. The fertilizing elements, nitrogen and phosphorus, will tend to remain in the water to a considerable extent, but quite often as cellular material of aquatic plants and animals, rather than being in the dissolved form. Those materials which can be expected to continue to exist relatively unchanged are the dissolved inorganic materials and the smaller sized fraction of the suspended particles.

The largest effect of urbanization will obviously be manifest in regions immediately adjacent to the Great Lakes themselves. This includes the more densely populated regions such as Chicago, Duluth, Milwaukee, Detroit, Toledo, Cleveland, and Buffalo. Thus it can be anticipated that urban drainage will significantly affect water quality in these areas. The

TABLE I

Pollution Inducing Materials

Category	Specific Parameter*	Possible Source
Bacterial	Total Coliform	Humans
	Fecal Coliform	Land Animals
	Fecal Streptococcus	Birds
	Pathogens	
Primary Nutrients	Nitrogen	Fertilizers
	Phosphorus	Decomposition of Organic Matter
		Leaching from Minerals
Organic Substances	Biochemical Oxygen Demand (BOD)	Humans and other Animals
	Chemical Oxygen Demand (COD)	Leaves and Grass Clippings
	Total Organic Carbon (TOC)	Oil and Grease
Inorganic Substances	Total Suspended Solids (TSS)	Erosion
		Dust and Dirt
		Earthen Channels
	Total Dissolved Solids (TDS):	
	Chlorides	Leaching from Minerals
	Carbonates	Street Deicing
	Sulfides	
	Heavy Metals	Industrial Refuse
Pesticides and Herbicides		Application within Area

*The concentration ranges experienced in combined sewer overflows and urban stormwater runoff for these parameters are shown in Table I-A.

TABLE I-A

URBAN DRAINAGE CHARACTERISTICS[5]

Parameter	Combined Sewer Overflow	Urban Stormwater
BOD_5 (mg/l)	30 - 600	1 - 700
COD (mg/l)		5 - 3,100
TSS (mg/l)	20 - 1,700	2 - 11,300
TS (mg/l)	150 - 2,300	450 - 14,600
Vol. TS (mg/l)	15 - 820	12 - 1,600
Settleable Solids (mg/l)	2 - 1,550	0.5 - 5,400
Organic N (mg/l)	1.5 - 33.1	0.1 - 16
Ammonia N (mg/l)	0.1 - 12.5	0.1 - 2.5
Soluble PO_4 (mg/l	0.1 - 6.2	0.1 - 10
Total PO_4 (mg/l)		0.1 125
Chlorides (mg/l)		2 - 25,000
Oils (mg/l)		0 - 110
Phenols (mg/l)		0 - 0.2
Lead (mg/l)		0 - 1.9
Total Coliforms (no./ 100 ml)	2×10^4 - 9×10^7	200 - 1.46×10^8
Fecal Coliforms (no./ 100 ml)	2×10^4 - 1.7×10^7	55 - 1.12×10^8
Fecal Strep. (no./100 ml)	2×10^4 - 2×10^6	200 - 1.2×10^6

direct effect of urbanization in other areas on the connecting waters of the Great Lakes is not well defined at the present time, although the potential for harmful effects obviously exists. The total impact of these areas, taken in the aggregate, may well be significant, especially with respect to conservative materials (chlorides, sulfides, heavy metals, etc.) and primary nutrients.

1.2 Summary

Residential and commercial-industrial land usage has exerted a significant deleterious affect on the water quality of the Great Lakes. This has been identified primarily in the surface water quality in the vicinity of the major population concentrations. This is true in certain waters of the Great Lakes themselves, notably southern Lake Michigan, the Detroit River, Lake Erie, Lake Ontario, Green Bay, and Saginaw Bay. It is also true for a large number of tributary rivers, such as the Fox River in Wisconsin; the Grand, Rouge and Saginaw Rivers in Michigan, the Maumee and Cuyahoga Rivers in Ohio; and the Buffalo River in New York.

These affects were originally felt to be primarily the result of point source discharges of untreated or inadequately treated domestic and industrial wastewaters. In the late 1940's the influence of recurring overflows of combined sanitary-storm sewers was also shown to be one of the primary factors in water quality degradation. During the last decade the more subtle effect of separate storm drainage from such types of land use areas has been studied.

It is not possible at the present time to provide a general quantitative evaluation of the affect of urban land drainage alone on water quality, because further efforts must be made on the abatement of point-source pollution from municipal waste treatment plants and industries. These effluents tend to mask the effects of land drainage, preventing the quantitative evaluation of such effects alone.

As adequate treatment is applied to the point source waste discharges, and the proportion of urban land usage within the Great Lakes watershed increases, the importance of land drainage sources of many pollutants will be magnified. Studies have been, and are continuing to be, made on the isolated effects of urban land drainage on water quality, and on methods to minimize any deleterious effects. Present studies have provided indications as to the seriousness of these drainage problems and have presented possible treatment schemes for abating pollution from such sources. Additional efforts should be made to further define the impact of these discharges, particularly in the Great Lakes area, and to optimize treatment technology with respect to urban drainage. Comprehensive studies into the pollutional effects of this drainage may also allow for the identification of new water management strategies for the protection of the Great Lakes.

1.3 Recommendations

In spite of the relatively large amount of work which has been performed in the area of urban runoff and combined sewer overflows, a great deal of information needs to be obtained on the effect of residential, commercial, and industrial land usage on water quality. This is particularly true in the Great Lakes Basin, where the quality of the water is of such critical importance to the health and economy of the region, and yet where so little quantitative information is available on what affect land usage has on this water quality.

The first priority would appear to be the delineation of land usage patterns within the Great Lakes Basin. Ideally, this would be performed for each drainage area within the various river basins which serve to transport water to the Great Lakes. This would allow for the verification and/or modification of newly developed relationships and analytical techniques described later for the conditions found within the Basin.

Performing such an analysis on this basin, however would not be practicable with respect to both time available and economic limitations. A logical compromise would be to perform such detailed analysis on selected drainage areas throughout the basin and then develop the technological tools for delineating land usage - water quality relationships for this geographical area. This would then allow for projections of water quality relationships on the basis of land usage defined for larger areas, preferably major river basins.

In addition to the delineation of land usage patterns in the Basin, further emphasis must be placed on obtaining water quality data, particularly with respect to quality modification due to drainage resulting from storm events. Such information is necessary for the development of runoff-water quality relationships for this area, and is currently nonexistent except for a relatively small amount pertaining to combined sewer overflows. A comprehensive program to obtain such data should be formulated in such a way that functions can be generated for specific types of land usage, various topographic features, and different densities of activities.

Since the previously performed studies have clearly shown that urban land usage has a significant deleterious affect on water quality, a third area of research interest is methods of minimizing the pollutional effect of such runoff through more effective control and/or treatment. This effort would include not only the various techniques which might be applicable, but also the cost-benefit relationships which exist for these various techniques. Particular emphasis should be placed on delineating pollution abatement measures which can be incorporated with other runoff control procedures. For example, flood control projects could be designed to allow for treatment of storm runoff prior to discharge to the receiving stream. Likewise, recreational projects could incorporate runoff retention basins which would allow for storage of runoff to

allow for natural purification as well as providing a recreational and aesthetic advantage. A high priority should also be placed on preventing the entrance of pollutants into the storm drain system. Examples of this type of effort would be the development and utilization of more effective street cleaning and deicing techniques.

A final area of needed investigation is a delineation of legislative and institutional arrangements existing or needed with respect to pollution resulting from urban runoff. Since the boundaries of governmental units do not in general coincide with drainage basin or river basin boundaries, the most cost effective means for reducing the effect of urban runoff on water quality may require the use of controls which apply to more than one governmental unit. Existing legislative authority for accomplishing such projects should be clearly defined, and the possibility of enhancing the potential for such interagency cooperation should be investigated and instituted where possible.

Section 2
SUPPORTING MATERIAL

2.1 Introduction

The affect of urban land usage on water quality is a combination of easily observed, gross effects (e.g. high coliform concentrations due to combined sewer overflows) and more subtle, relatively undefined effects (e.g. increase air pollution, which is "washed out" by precipitation). The most significant and wide spread effects are on the surface waters of the Great Lakes and their tributaries, however the potential for ground water contamination and eventual introduction to the Great Lakes via this pathway exists. It is difficult to assess the specific effects of this type of drainage, since they are generally obscurred by the effects of the discharge of untreated or inadequately treated municipal and industrial wastewaters. During the past decade, however, studies on isolated units of urban areas have provided indications of the possible consequences of urban drainage alone on water quality.

2.2 Scope of Study

The basis of the present discussion was a thorough review of the literature,concentrating primarily on the period since 1960. The various pertinent literature abstract, specifically Water Resources Abstracts and Engineering Index were reviewed for information sources. In addition, the annual indices of various professional journals were surveyed for pertinent information. The primary journal sources included the following:

Water Pollution Control Federation Journal
Environmental Engineering Division Journal, A.S.C.E.
Irrigation and Drainage Division Journal, A.S.C.E.
Water Resources Research
Water Resources Bulletin
Pollution Engineering

Water and Sewage Works
Ground Water
Water Well Journal
Public Works

The primary source of information, however, was various publications of the U. S. Environmental Protection Agency and its forerunners.

Section 3
PREVIOUS STUDIES

3.1 Pollution Transport in Urban Areas

The pollutional characteristics of land drainage will be dependent to a great degree on the means of transport from the point of incidence to the receiving water. It must be recognized that the precipitation itself may become somewhat polluted as it falls to the earth, due to contact between the precipitation and air borne pollutants. This will be especially true in areas of heavy industrial activity. Weibel, et al[1], presented data on pollutant concentrations in rainfall in both urban and rural locations of Ohio. The average concentrations of selected constituents for a number of storm events is shown in Table II.

Table II

Constituent	Urban	Rural
Suspended Solids (mg/l)	13	11.7
COD (mg/l)	16	9.0
Nitrogen (mg/l as N)	1.27	1.17
Organic Nitrogen (mg/l as N)	0.58	0.31
Inorganic Nitrogen (NO_3, NO_2, and NH_3)	0.69	0.86
Hydrolyzable Phosphate (mg/l as PO_4)	0.24	0.08
Organic Chlorine (µg/l)	0.28	0.22

It can be seen from these results that a somewhat higher contamination can be expected to occur in urbanized areas. Further evidence of nutrient input resulting from precipitation was cited by Aulenbach and Clesceri[2]. Studies which they discussed indicated nitrogen concentrations in rainwater ranging from 0.1 to 2 mg/l and phosphorus concentration as high as 10 µg/l. Although no direct identification of the source of these materials was made, it is well known that

oxide of nitrogen are contributed by most combustion processes, and should thus be more significant in urban areas due to increased levels of heating and automative exhausts. It must be emphasized that this pollution resulting from the "scrubbing" of atmospheric contaminants may occur at considerable distances from the point at which the contaminants are originally generated. Thus, although the generation of air pollution will be higher in urban areas, the affect on water quality may be exerted at considerable distances from these areas as a result of transportation by wind action.

Drainage resulting from precipitation on urban land areas will generally encounter a degree of overland flow prior to entering a sewer or receiving water. Depending upon the types of surfaces encountered (lawns, parking lots, streets and gutters, etc.) this runoff can become contaminated by numerous materials such as fertilizers applied to lawns, grease and oil from parking areas, dust and dirt accumulations on roofs and streets, fecal materials from birds and animals, pesticides and herbicides applied to vegetation, salts applied for snow removal, and possibly others. The degree of contamination is dependant upon several factors, such as land topography, distance of overland flow, antecedent rainfall, and land management activities.

The majority of land drainage from residential and commercial-industrial areas will be transported from its location of generation to the receiving water by some type of drainage system. This system may be of either a combined storm and sanitary sewer system or a separate system of either the open ditch or closed conduit form. In the former, the sewer serves to carry the routinely generated sanitary and/or industrial wastes to the treatment plant during periods of dry weather, while during periods of wet weather, the storm drainage is carried along with the normal dry weather flow. Since the treatment plants are generally incapable of handling this greatly increased flow during storm periods, overflow devices allow the

discharge of the mixed storm and sanitary wastewater directly to the receiving water. Numerous studies have identified the significant pollution problem stemming from the overflow of combined sewers[4, 6, 7]. It was soon found during the investigation of overflow problems, however, that the storm drainage emanating from separate sewer systems was itself sufficiently polluted to cause deterioration of water quality in receiving bodies[1, 3, 8, 9, 10]

It is thus evident that a comprehensive quantitative evaluation of the affect of urban drainage on water quality requires a thorough evaluation of the particular region of concern with respect to transport of pollutants.

3.2 Affects of Land Usage on Water Quality

The earliest investigations into the affects of residential and commercial-industrial land usage on water quality have dealt primarily with the affect of urbanization on water quality degradation by point source discharge of industrial and municipal wastewaters. In the lates 1940's, the problem of recurring overflows from combined sanitary-storm sewers became a concern[11], here again primarily as a result of the relatively small amount of municipal waste discharged along with the much larger volume of storm drainage. Little definitive investigations were performed in this area until the early part of the last decade. In 1963, Palmer[12] discussed the feasibility of using combined sewers, indicating that the quality of stormwater flow in urban areas is sufficiently polluted even without the presence of sanitary wastewater to cause deleterious effects on receiving water quality. A limited amount of data on stormwater drainage alone, obtained from catch basins in the city of Detroit, showed coliform bacteria counts ranging from 2,300 to 430,000 per 100 ml. Based on these data, the author concluded that separation of the sewer system would not significantly lower the pollutional effect of overflows on the river system due to

the pollutional content of the stormwater drainage alone.

In 1964, Weibel et al[3] presented the results of a study of a small separate sewer system in Cincinnati. The area studied was primarily residential and commercial with a total area of 11 hectares (27 acres) of which approximately 37 percent was impermeable (pavements, sidewalks, building surfaces, etc.). The results obtained by sampling over 60 percent of the 240 hours of recorded runoff from July, 1962 through September, 1963 are shown in Table III.

In 1966, an update of the previous study was made[1] which presented projected comparisons between runoff from the study area during storm periods and with raw sewage from the same area. It was projected that urban runoff would contain 24 times the suspended solids load of raw sewage, 5.2 times the COD load, 1.1 times the BOD load, 0.7 times the phosphate load, and 2.0 times the total nitrogen load. It is thus obvious that this type of land usage can be detrimental to water quality during periods of storm flow, and must be considered in the formulation of master plans for water quality control.

The pollutional effects of both combined and separate sewer systems in southeastern Michigan were discussed in a series of articles[8, 9, 10]. These studies were concerned with analysis of composite drainage from relatively large areas - 1,540 hectares (3,800 acres) in Ann Arbor and 8,900 hectares (22,000 acres) in Detroit-which contained diverse types of urban land usage. The Conner Creek combined sewer system was comprised of 28 percent streets, 3.4 percent commercial and institutional, 10.3 percent recreational and open space, 19 percent industrial, and 39.3 percent residential areas. The Allen Creek separate storm drain in Ann Arbor was made up of 23.5 percent street, 15.7 percent commercial and industrial, 23.0 percent recreational and open space, 1.0 percent industrial, and 36.8 percent residential areas. A synopsis of the large amount of data gathered in this study is presented in Table IV[8].

Table III

Constituent Concentrations in Urban Runoff at a Cincinnati Sampling Point July 1962 through September 1963*[3]

Parameter	Range	Mean
	Units	
Turbidity	30 - 1,000	170
Color	10 - 380	81
pH	5.3 - 8.7	7.5

Parameter	Range	Mean
	mg/l	
Alkalinity	10 - 210	59
Calcium (as $CaCO_3$)	24 - 200	63
Magnesium (as $CaCO_3$)	2 - 46	15
Total Hardness (as $CaCO_3$)	29 - 240	78
Chloride	3 - 35	12
Suspended Solids	5 - 1,200	210
Volatile Suspended Solids	1 - 290	53
COD	20 - 610	99
BOD	2 - 84	19
Nitrite (as N)	0.02 - 0.2	0.05
Nitrate (as N)	0.1 - 1.5	0.4
Ammonia (as N)	0.1 - 1.9	0.6
Organic Nitrogen (as N)	0.2 - 4.8	1.7
Soluble Phosphate (as PO_4)	0.07 - 4.3	0.8

*January and February not included.

Table IV

Characteristics of Storm and Combined Sewer Overflows

Parameter	Separate System (Allen Creek Drain-Ann Arbor) Range			Mean	Combined System (Conner Creek-Detroit) Range			Mean
Total Coliform (#/100 ml)	26,500	-	17,500,000	1,200,000	495,000	-	90,000,000	9,400,000
Fecal Coliform (#/100 ml)	7,500	-	1,115,000	82,000	200,000	-	17,000,000	2,700,000
Fecal Strep. (#/100 ml)	13,800	-	730,000	140,000	295,000	-	1,570,000	580,000
Ammonia -N (mg/l as N)	0.27	-	0.68	0.48	0.00	-	11.52	3.25
Organic -N (mg/l as N)	0.00	-	0.52	0.36	0.08	-	1.09	0.37
Phenols (mg/l)	0.001	-	0.011		0.011	-	0.125	0.027
Suspended Solids (mg/l)	900	-	2,062	1,280	23	-	1,398	150
Total Phosphates	1.6	-	9.5	2.9	1.8	-	25.0	9.0

In addition to the magnitudes of the concentrations observed, it is important to note the fecal coliform to fecal streptococci ratio. Ratios greater than 2:1 are generally indicative of domestic wastewater pollution, while ratios less than 1:1 indicate pollution primarily derived from warm-blooded animals other than man[13]. The observed ratio in the combined system was approximately 4.7, while in the separate system it was 0.6.

Further investigations into bacterial pollution of land drainage was performed by Beldreich, et al, in 1968[13]. Significant contributions of bacterial pollution were traced to warm-blooded animals other than man. In the urban community, such animals would likely be pets - particularly cats and dogs - and a substantial rodent population. One analysis of storm drainage from a business district also contained 4,500 Salmonella organisms per 100 ml. These results are particularly significant with respect to the impact on bathing areas by intermittent releases of storm drainage in the nearby vicinity.

Additional work was performed in the Cincinnati test area discussed previously[14] where data from eight storm events resulted in BOD values of 4-220 mg/l, COD levels of 48-810 mg/l, and suspended solids concentrations of 110-930 mg/l. The study indicated that 2 to 6 mg/l of chlorine applied for 20 minutes effected a 99.99 percent kill of total coliforms, fecal coliforms, and fecal streptococci. An aftergrowth of total coliforms was seen to occur within 24 to 72 hours, however, there was no significant aftergrowth of the other two species. These results indicate the importance of using fecal coliforms rather than total coliforms as a microbial indicator of pollution.

One of the more comprehensive investigations into the influence of residential and commercial-industrial land usage on water quality was performed by the American Public Works Association[15]. This project involved the study of 18 separate areas in Chicago for the affect of street litter, catch basins, and air pollution.

The BOD of street litter was found to exert 3 to 14 mg BOD per gram of dry material with an average of 5 mg/l. Catch basins were found to be a source of shock pollution, since the liquid in them tended to become septic between storms, and BOD values of 60 mg/l for this material was common. Street litter was found to create a water pollution potential when it comes in contact with runoff waters. The amount of litter deposited from various sources were determined from these test areas to vary from 0.5 to 8 pounds per day per 100 feet of curb. The average values showed 4.7 pounds/day/100 feet of curb for commercial areas, 3.5 pounds/day/100 feet of curb for multiple family areas, and 2.4 pounds/day/100 feet of curb for single family residential areas. The most significant component of the litter with respect to water pollution was the dust and dirt fraction (less than 1/8 inch diameter) which amounted to 0.4 - 5.2 pounds/day/100 feet of curb. The average pollution contaminants contained in this fraction were as follows:

BOD	5	mg/g
COD	40	mg/g
Nitrogen	0.48	mg/g
Phosphates	<0.05	mg/g
Total Bacteria	$>10^7$	/g
Coliforms	$>10^6$	/g
Fecal Enterococci	5400	/g

The most determinable measure of the pollutional potential of street litter was deemed to be the BOD of the soluble dust and dirt fraction. Comparing the results obtained in this study with normal sanitary sewage, it was projected that the street litter BOD would be equivalent to the raw sewage of five persons per day per mile. This is a pollution potential of one percent of the raw sewage loading or five percent of the loading anticipated for the effluent of a secondary treatment facility. Projecting this into a time frame normally associated with storm events, the shock loading on receiving waters per mile of

street could be 160 percent of the raw sewage BOD and 800 percent of the secondary treatment effluent during a two hour runoff period for a 14 day accumulation of material. The basic conclusion of this study was that proper urban "house-keeping", including effective street cleaning operations, could significantly reduce the affect of storm drainage on water quality.

Several other studies have been performed on widely scattered communities throughout the country, including Bucyrus, Ohio, New Orleans[16], Tulsa[17], Atlanta[18], Washington D.C.[19], Sacramento[20], and Durham, N.C.[21]. Even though the majority of these studies are concerned with communities remote from the Great Lakes area, the information derived from them can be applied, with proper modifications, to communities within the Great Lakes Basin. Representative data for separate storm sewers as determined from several of these studies are presented in Table V.

Several recent studies have been concerned with specific aspects of the pollutional character of urban drainage. Two recent reports by the URS Research Company have been concerned specifically with street surface contaminants. The first with water pollution aspects in general[22] and the second with toxic materials in particular[23]. It was determined in these studies that street surface runoff is highly contaminated, contributing considerably more pollution load than the sanitary sewage system during the period of runoff. Other studies have been concerned with the affects of street salting on water quality[24]. It was found that not only are high concentrations of salts present in highway runoff, but special additives, such as mercury, may present future problems due to their severe latent toxic properties.

It can easily be seen from the previously discussed studies that the usage of land for residential, commercial, and industrial purposes will tend to have a significant affect on water quality. It is thus necessary to provide the proper

drainage management and control procedures to these areas to minimize the deleterious effect of urban runoff on the receiving waters.

Table V

Average Concentrations of Selected Parameters from Storm Sewer Systems

	Durham, N.C.[21]	Tulsa, Okla.[17]	Atlanta, GA[18]	Wash. DC[19]
BOD (mg/l)	14.5	11.8	18	19
COD (mg/l)	179	85.5	60	335
TSS (mg/l)	-	367	-	1,697
Org-N (mg/l)	-	0.85	-	-
NH_3-N (mg/l)	-	-	-	-
PO_4 (mg/l)	0.6	1.15	0.8	1.3
Total Coliform (#/100 ml)	-	87,000	-	600,000
Fecal Coliform (#/100 ml)	30,000	420	-	310,000
Fecal Strep (#/100 ml)	-	6,000	-	21,000

3.3 Effects of Urban Drainage On Water Quality of the Great Lakes

There has been very little investigation into the effects of residential and/or commercial-industrial land usage on the water quality of the Great Lakes. Those studies from which pertinent information can be derived are primarily those which are concerned with combined sewer overflows.

A study was performed on a 200 hectare (495 acre) site in Milwaukee comprised primarily of one and two family residential dwellings with a small amount of commercial area[25]. The area has approximately 42 percent impervious area representing a high density situation. Fifty-five overflows were monitored over an eighteen month period from May, 1969 through November, 1970. Although flow rates were not measured, thereby preventing the calculation of total pollutant load from this area, the con-

centrations of various parameters are shown in Table VI.

Table VI

Quality of Combined Sewer Overflow - Milwaukee, Wisconsin (1969-1970)[25]

Parameter	Concentration (mg/l)*
COD	161 ± 19
BOD	49 ± 10
Total Solids	378 ± 46
Total Volatile Solids	185 ± 23
Suspended Solids	166 ± 26
Volatile Suspended Solids	90 ± 14
Total Nitrogen	5.5 ± 0.8
pH (Units)	7.2 ± 0.1
Coliform Density(per ml)	(62.5 ± 27) x 10^3

*95 percent confidence level range

Of equal importance in this study was the analysis of the first flush phenomena, which was observed on 12 overflows. The characterization of the waste during these events is presented in Table VII.

Table VII

Quality of First Flush of Combined Sewer Overflow - Milwaukee, Wisconsin (1969-1970)[25]

Parameter	Concentration (mg/l)*
COD	581 ± 92
BOD	186 ± 40
Total Solids	861 ± 117
Total Volatile Solids	489 ± 83
Suspended Solids	522 ± 150
Volatile Suspended Solids	308 ± 83
Total Nitrogen	17.6 ± 3.1
Ortho Phosphate	2.7 ± 1.0
pH (Units)	7.0 ± 0.1
Coliform Density(per ml)	(142 ± 108) x 10^3

* 95 percent confidence level range

Comparison between these two tables shows that the first flush, which may last for 20 to 70 minutes, exhibits pollutant concentrations two to four times those in the extended overflow. The concentration levels in these first flushes are also observed to be of the same order of magnitude of raw domestic waste.

The Southerly Sewerage District of Cleveland consists of residential, commercial, and industrial areas covering approximately 25,000 hectares (62,000 acres) and an estimated 600,000 people. A 1972 report[26] indicates that, while the treatment facility has a hydraulic capacity of 7 cubic meters per second (160 mgd), the wet weather flow ranges from 7 to 35 cubic meters per second (160 to 800 mgd), causing a substantial discharge of combined sewer overflow to the Cuyahoga River and Lake Erie. The report concluded that a total of about 52 overflows a year, generally lasting for five to six hours, can be expected from this area. The concentrations of various parameters evaluated in the study were observed to vary significantly between summer/fall overflows (late July - early November, 1970) and spring overflows (May - early June, 1971). Representative results obtained are shown in Table VIII.

Table VIII

Quality Characteristics of Combined Sewer Overflow - Cleveland, Ohio (1970-1971)[26]

	Summer/Fall		Spring	
Parameter (mg/l)	Mean	Range	Mean	Range
BOD	92	16 - 580	171	85 - 245
COD	308	57 - 711	462	196 - 759
Total Solids	590	264 - 1,238	814	532 - 1,275
Suspended Solids	234	28 - 1,560	411	177 - 976
Settleable Solids (mg/l)	5.26	0.2 - 19	6.98	1.5 - 14
Fecal Coliforms (10^6/100 ml)	4.55*	0.09- 49	0.53	0.06 - 2.00

*median

Characteristics of the dry weather flow at the treatment plant as determined hourly over an 18 hour period from 28 October 1970 are shown in Table IX.

Table IX

Quality Characteristics of Dry Weather Flow - Cleveland, Ohio (1970-1971)[26]

Parameter (mg/1)	Mean	Range
BOD	130	60 - 185
COD	383	164 - 543
Total Solids	634	441 - 820
Suspended Solids	192	64 - 249
Settleable Solids (ml/1)	5.3	0.5 - 9.5
Fecal Coliforms (10^6/100 ml)	15.7*	5 - 62

*median

Thus, the overflow characteristics are essentially the same as raw domestic sewage, except suspended solids are considerably higher while fecal coliforms are lower. Using the average of the range of wet weather flows assumed, this means the equivalent of nearly 284 million liters (75 million gallons) of raw waste would be discharged approximately 52 times per year from this area.

Comparing the data from these two communities, it is obvious that the waste characteristics in one area may differ significantly from those in another area. It is also shown in the Cleveland situation that the pollutional load for a given area will tend to vary with seasons of the year. The state-of-the-art at the present times does not allow for an apriori evaluation of the effect of urban drainage on water quality. It must also be emphasized that the concentration of pollutants alone or even total pound loadings, will not provide information as to the effect of such discharge on water quality.

The location of the drainage or overflow outlet with respect to the ultimate receiving water must be taken into consideration as well. Discharge into bay areas, where the water tends to be more quiescent, will have greater localized effect than discharges into areas with high current velocity. A region such as Green Bay, for example, exhibits a greater problem with respect to pollution (although the effect of urban drainage has not been isolated from the effect of point-source discharges) than the upper Detroit River, where the Conner Creek combined system overflows.

The upper Detroit River is one area of the Great Lakes where data are available to show the actual effect of urban drainage on water quality. One phase of the Public Health Service investigation of the Detroit River involved a special study on combined sewer overflows in the upper river[27]. The following conclusions were drawn from this study.

1. Coliform, fecal coliform, and fecal streptococcus densities increased in the Detroit River, following an overflow from combined sewers, 10 to 50 times over the values found during dry weather.
2. Coliform densities in the Detroit River following an overflow often exceeded 300,000 per 100 ml and at times exceeded 700,000 per 100 ml.
3. All high bacteriological values in the Detroit River during or following an overflow were found below Conners Creek. Bacteriological densities above this point stayed fairly constant during wet and dry conditions. Conners Creek represents the most upstream location of many combined sewer outfalls which extend to the mouth of the River.
4. Analysis of the City of Detroit sampling records reveals individual analyses exceeding 800,000 coliforms per 100 ml in the Detroit River on the day following significant rainfall.
5. High bacteriological densities following overflows were found at both the City of Wyandotte water intake and the new City of Detroit intake near Fighting Island. The Wyandotte values exceeded 100,000 per 100 ml and the Fighting Island values 10,000 per 100 ml.

6. The effect of overflows on water quality in the Detroit River has been observed as long as four days after the rain subsided.

7. Each of the nine storms individually investigated produced a severe effect on water quality in the Detroit River as evidenced by increased bacterial contamination. This effect was also noticed in statistical evaluation of regular data by wet or dry conditions.

8. The length of the effect of overflows of combined sewers upon water quality in the Detroit River varies from one to four days after the beginning of the actual discharge.

9. The greater the rain the longer the period of overflow and more severe the effect on the Detroit River.

10. While bacteriological analysis was used to compare normal conditions with those found during or following an overflow, other observations were made by field personnel in the area during heavy rains which indicated the deleterious effect of the overflows upon water quality in the River. Field notes on these occasions described debris and garbage as well as excrement floating down the Detroit River.

11. Analysis of rainfall, overflow, and stream quality records reveals that during a nine-month period in 1963 (March - November) overflows from combined sewers affected water quality in the Detroit River during part or all of 88 days. This represents 32 percent of the days in the nine-month period. This phenomenon occurred during the year of lowest accumulated rainfall and could represent an even greater effect on Detroit River water quality during a year of normal rainfall.

Routine water quality monitoring does not necessarily provide information on the effect of urban drainage on water quality, since the conditions producing the runoff (i.e. precipitation) do not present the most ideal circumstances for sampling. Nonetheless, more recent data obtained on the Detroit River by various state and federal agencies indicate the quality of the water still deteriorates between Belle Isle and the Rouge River, due primarily to combined sewer overflows.

3.4 Assessment and Quantification of Urban Runoff

Although, as has been discussed previously, there has been a considerable effort into documenting the characteristics of urban runoff in general and its potential for polluting the receiving streams, relatively little has been accomplished in the area of assessment of pollution in storm water as it relates to specific land activities. The most comprehensive work to date in this area was performed by Avco Economic Systems Corporation in Tulsa, Oklahoma[17]. The output of this study included functional relationships for estimating the concentrations of certain pollutant parameters based on either precipitation variables or land usage. The study concludes that these functional relationships can be used to obtain a first order estimate of the average pollutant concentrations in urban watersheds at geographical locations other than Tulsa, however, until these relationships are verified for various areas within the Great Lakes Basin they should be applied here only with caution. This study's primary importance lies in providing a format which should allow for the maximum information on storm water pollution from a given urban area to be obtained at a minimum effort.

This report contains a large number of regression equations which attempt to correlate various parameters with the concentration of certain pollutants under various urban land usages. For example univariate equations are presented for BOD concentrations from a residential area as a function of environmental condition, residential density, ratio of covered sewer to the total length of main channel, or percentage of non-arterial streets. Multivariate realtionships are presented correlating this parameter with various combinations: environmental condition, covered sewer ratio, and topographic conditions; environmental conditions, covered sewer ratio, percentage of arterial streets and percentage of non-arterial streets; or covered sewer ratio and percentage non-arterial streets. These seven equations can thus be tested using the conditions existing in a

given area and runoff data obtained for that area to determine which formulation best describes the area under study. Although these relationships provide a framework for analyzing a particular area, considerable effort must be expended to provide the necessary input data, and evaluate the reliability of the results obtained.

Other recent activities in this area have involved the development of comprehensive mathematical models using a theoretical approach, with the coefficients determined by and checked against the available data[20, 28]. Recent studies with such an approach show much promise in evaluating the runoff characteristics from urban areas, which can then be integrated with currently available receiving water models to delineate the actual effect of the runoff on water quality. Although this approach holds great promise, a great deal of development work is still required to authenticate is general applicability.

It must be stressed that the present state-of-the-art does not allow for apriori quantitative assessment of pollutant loading from various urban land usages. This is well illustrated by the Avco study in Tulsa[17] which determined average BOD loadings from two ostensibly identical, medium density residential areas to range from 16 to 39 kg/ha/year (14 to 35 pounds/acre/year). These same two areas exhibited total solids loading of 1325 and 553/kg/ha/year (1183 and 494 pounds/acre/year), respectively.

With regards to the Great Lakes in total, it is impossible to provide a meaningful total loading factor. In addition to the limitations discussed above with respect to quantifying pollutant loads from urban activity, the greatest amount of information presently available deals with non-conservative materials (degradable organics, nutrients, bacteria, etc.) which is of primary significance only in relatively localized areas. To consider such pollutant loads from urban land drainage for the Great Lakes as a whole is without meaning.

The available information on characteristics of urban land drainage does, however, allow for an estimation of the relative significance of this particular activity as compared to other activities with respect to a unit area basis. The loading function ranges to be expected for various land usages, assuming the information obtained for the Tulsa region[17] will hold true in the Great Lakes region, even though climatic differences between the two areas are significant, are shown in Table X. Data from the same study, included in Table X, shows that the pollutant loadings from relatively open land (airport and park land) are generally much lower than any type of urban usage. In contrast, assuming a residential population density of 49.4/ha (20/acre), and treated wastewater effluent with a BOD concentration of 10 mg/l and phosphorus concentration of 1 mg/l, the point source pollutant loading would be 68 kg/ha/year (61 pounds/acre/year) of BOD and 6.7 kg/ha/year (6 pounds/acre/year) of phosphorus. This is generally much higher than the non-point source pollutant loadings. A great deal of additional information with regards to such loading functions for the Great Lakes area must be obtained before a more accurate assessment can be made.

In general, the state-of-the-art with respect to determining the effect of residential and/or commercial-industrial land usage is best summed up by the first conclusion presented in an urban runoff study in Sacramento[20].

> "The necessary data are not available in the Study Area or most areas to determine or predict distributions of storm water runoff and combined sewage flows and pollutant contents, and distributions of the corresponding receiving water characteristics. Most, if not all, of the characterizations of storm water runoff and combined sewage that has been performed and reported is totally unsuitable for predicting in a quantitative manner the storm water runoff. The best that these data can provide are qualitative indications of the magnitude and ranges of expected variations in the characteristics."

Table X

Pollutant Loading Ranges for Various Urban Land Usages

Land Usage	Pollutant Loading (pounds per acre per year)*				
	BOD	COD	Kjel. Nitrogen	Soluble Orthosphate	Total Solids
Residential	14-25	90-290	0.8-2.6	1.3-3.3	490-1,400
Industrial	21-44	160-320	1.1-3.0	1.5-8.0	600-5,100
Commercial	27-48	150-470	3.3-3.6	2.9-3.1	920-1,900
Airport	25	140	1.2	1.7	630
Recreational	12	60	1.1	1.1	660

* Conversion -- 1.12 times pounds per acre per year = kilograms per hectare per year

3.5 Institutional Arrangements for Control of Pollution from Urban Land Use

Several governmental mechanisms exist for the minimization of the affect of urban land usage on water quality. The greatest problem in this regard at present is providing the implementation of existing mechanisms. Zoning ordinances, which are of almost universal existance in urban areas, provide a ready means for minimizing many pollution problems. Provisions for open spaces, although tending to increase urban sprawl, will tend to minimize localized inputs of heavy pollution loads. These same ordinances can be used to prevent the location of refuse disposal facilities and open storage areas in regions more susceptible to leaking of pollutants from overland and/or subsurface flows.

A second legislative area of prime importance in this regard is erosion control ordinances. Soil erosion resulting from overland drainage, particularly during construction operations, are known to have a significant effect on water quality. Erosion control ordinances, similar to that recently enacted by the State of Michigan, will greatly reduce the impact of stormwater drainage from urban areas if stringently enforced. Such ordinances are not overwhelmingly in existence at the present time, and a great deal more input of personnel is required in those areas where such ordinances do exist.

The most effective means for pollution control is prevention of pollutant generation. There are several actions in this regard which can be taken in the Great Lakes Basin to reduce the impact of urban land usage on water quality. One obvious item is more careful management of snow removal practices. Indiscriminate use of salt and/or sand can result in unnecessarily high loadings of chloride and suspended solids to the receiving waters. Similarly, careful selection and application of pesticides and herbicides can reduce the effects resulting from this activity. This aspect can best be treated by educational

programs involving governmental personnel, so that they may perform the necessary functions of environmental maintainence with greater regard for the potential pollution effects.

Section 4

TECHNOLOGY FOR CONTROL OF URBAN RUNOFF

Reviewing the sources of water pollution from urban runoff, it is obvious that there exists two basic approaches to minimizing the effects of such drainage on water quality. The first approach is to control the entrance of polluting material to the channels draining the area; the second is to provide treatment to the drainage prior to its release to the ultimate receiving body of water. As in other areas of wastewater management, proper control will generally be less expensive than treatment. This is especially true in newly developing areas, where control procedures can be instituted at less cost than attempting to provide controls in existing systems.

The most obvious and most widely studied means of controlling the effects of drainage on water quality is the use of separate storm and sanitary sewers rather than combined systems. Although it has been shown that separation of sewers alone will not solve the pollution problems resulting from urban land activity, the concentrations of pollutants discharged during combined sewer overflows may be two to three times higher than would be the case for storm sewer discharge alone[18, 19, 20, 29] Such combined systems are relatively common in the Great Lakes area, as was indicated in a report on combined sewers by the American Public Works Association[6]. This report, which surveyed only United States communities, indicated that 65 percent of the population of the western Great Lakes Basin of the United States (Minnesota, Wisconsin, and Michigan portions) were served by combined sewers, while 44 percent of the United States population of the Lake Erie Basin were served by combined systems. This represented a population of approximately 4.4 million people based on the population of these areas at the time of the study. The majority of people served by these systems resided in larger communities.

Control of combined sewer overflows is generally accomplished by optimization of the existing system. This can take the form of maximizing the volume of waste treated at the sanitary plant during the early period of the storm when pollutant concentrations are generally higher, improving overflow regulator maintenance to provide more carrying capacity in the sewer and preventing overflows during dry periods, and general "housekeeping" of the system. Control may also be provided by storage systems to contain at least a portion of the wet weather flow, which can then be diverted to the treatment facility after the passage of the storm.

The intermittent and random nature of combined sewer overflows makes the use of conventional biological treatment processes generally unsatisfactory. Many new approaches to treatment are being investigated, however, generally under the auspices of the Storm and Combined Sewer Pollution Control Program of the United States Environmental Protection Agency[7]. Methods currently under investigation include:

1. Fine-mesh screening and microscreening
2. Dissolved air flotation
3. Rotating biological contactors
4. High-rate plastic media trickling filters
5. High-rate, single -, and dual - media filtration
6. Vortex, swirl, and helical separators
7. Advanced disinfection methods
8. Tube settlers
9. Powdered and granular activated carbon adsorption
10. Polymer and other chemical additives for improved settling, microscreening, filtration, and flotation
11. Chemical oxidation
12. In-line or in-sewer treatment

Communities within the Great Lakes Basin where projects have been or are being carried out include the following[7].

Detroit:	System monitoring and remote control Polymeric additives Rainfall/runoff relationship definition
Milwaukee:	Chlorination Rotating biological contactor Screening, chemical oxidation, and split flow air flotation
Cleveland:	High rate filtration Hypochlorination, screening, generally improved maintenance
Sandusky, OH	Underwater temporary storage
Mt. Clemens, MI	Lagoon treatment and reuse
Bucyrus, OH	Storage and treatment
Akron, OH	Underground storage/treatment
Kenosha, WS	High-rate biological treatment

The Detroit monitoring system has shown the feasibility of increasing the pumping rate at the treatment plant at the onset of storms as monitored by remotely stationed rain gauges. This allows for greater sewer storage capacity and reduces the incidence of overflows[30]. The feasibility of off-stream storage and subsequent treatment was shown in studies at two Ohio communities[29, 31]. Studies on external treatment processes also exhibited the feasibility of this approach to combined sewer overflow abatement. Studies in Milwaukee[32] have shown that screening/flotation can effectively reduce pollution from combined sewer overflows. Effluent from the system was generally comparable to secondary effluent. The projected capital costs for a system to treat the once-in-five-year storm was $21,000 per mgd capacity or $3,828 per acre served. Ultra-high-rate filters have been studied at Cleveland[26]. Utilizing screens and high rate filters with polyelectrolyte addition, average suspended solids removal of 93 percent and BOD removal of 65 percent were obtained. Estimated capital cost for this process were $23,000 per mgd, which is essentially the same as the costs for the screening/flotation process used in Milwaukee. Higher levels of treatment were shown feasible

in a study utilizing powdered activated carbon, alum and polymer addition, and filtration[33]. Average removals in excess of 94 percnet of BOD and COD and 99 percent of suspended solids were achieved on combined sewage, although the capital costs were considerably higher than the previously discussed systems.

Specific investigations into the management, control, and/or treatment of stormwater runoff alone has been far less extensive than is the case with combined sewer overflows. However, a recent system study was conducted to determine the technical and economic feasibility of using small storage reservoirs as a means of storm water pollution control for an urban area[34]. Based on this study, it was determined that the use of small storage basins dispersed throughout an urban community for the control of storm water pollution was technically feasible and economically attractive compared to other alternatives. Furthermore, the storm water collected in these basins could be treated to provide approximately half the water demands of a typical residential community. Thus the production of potable water for domestic supplies or non-potable water for industrial usage may serve to recover at least a portion of the expense involved in controlling the effect of urban drainage on water quality.

The above mentioned studies, and many others being performed across the country[5], have provided evidence that the technology exists for effective management and control of combined sewer overflows as well as urban stormwater runoff. The controlling element in this, as in all aspects of pollution control, is the economical considerations. Justification of such expenditures requires the accurate delineation of the consequences on water quality if urban drainage is discharged without treatment.

Section 5

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Category A3 - Transportation

TRANSPORTATION AREAS

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To be used as portion of the U. S. Task A Report on
GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by
the International Joint Commission - prepared in
partial fulfillment of U. S. Environmental Protection
Agency Contract No. 68-01-1598

September 1974

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Section 1

SUMMARY

1.1 Introduction

Land use activities related to the field of transportation include: interstate, state and county highways and roads; railroads; airports; and pipeline and utility corridors. Runoff from the surfaces of these facilities during their construction, operation and maintenance carries many different classes of pollutants to nearby streams.

Sediments from soils eroded during site clearing and construction along rights-of-way; salts used as deicing chemicals; oils, heavy metals, and other waste products from vehicle emissions and highway operations; and roadside litter and spilled materials constitute the types of pollutants from transportation activities.

1.2 Summary of Findings and Conclusions

The impact that various land use activities associated with transportation have on the quality of the boundary waters of the Great Lakes Basin is difficult to assess quantitatively because of the limited amount of information that is currently available. The sources of pollution and the types of pollutants have generally been identified; but more field data must be collected, and additional research and development work must be completed before the full extent of the problem can be determined. However, there are a significant number of roads, railroads, airports and utility lines, which in the aggregate may form significant non-point sources of pollution to waters in the Basin.

The provisions of the National Environmental Policy Act of 1969 require that an environmental impact statement be prepared for all projects that affect the environment and are supported by federal funds. In order to provide the type of information required for these reports, research studies on highway activities that affect water quality have been initiated. However, few results from the studies on the other activities in the transportation area are available. The requirement for environmental impact statements has also lead to a reduction in the pollution from highway activities by defining potential problems and requiring methods to control them.

Most states in the region attempt to regulate non-point pollution from highways, railroads and utility lines under general legislation directed at pollution control. However, a few states have developed specific programs to control sedimentation and limit herbicide usage, especially where these activities may impact on nearby surface waters or adversely affect groundwater quality.

Other states provide information about methods that can be used for the temporary impoundment of surface runoff at highway construction sites to prevent siltation in water courses. However, all of these efforts are directed at controlling sediment transport at new construction activities; and have little effect at existing highway operations, where erosion may still result in significant sediment loadings in nearby streams.

Most of the state and county highway departments in the Basin have adopted a "bare pavement" policy with regards to snow and ice removal from road surfaces; but there are few state policies to control the potential pollution from this type of activity. There are few alternatives to salting roads in the winter time to prevent icing, but most highway departments do attempt to optimize salting practices to limit deleterious effects.

While transportation activities within the Basin are expected to continue to increase with the continued development of the region, much of this growth will occur under improved environmental policies and regulatory controls. In addition, this growth may be largely accomplished through improvements in existing facilities and not through whole new systems. Thus, future impacts from land uses related to transportation activities could be less serious than those that existed in the past.

1.3 Recommendations

It is not possible to provide a quantitative assessment of the impact that land use activities in the transportation category have on water quality in the Great Lakes Basin because the information to make the assessment is not available at the present time. Much more research aimed at establishing the rate of production of, and the amount of waste products generated by all activities in the transportation sector must be undertaken. Of particular interest is the pollution from the construction, operation and maintenance of state and county highways and roads, railroads, airports and utility lines.

Section 2

SUPPORTING MATERIAL

2.1 Introduction

Transportation as a category of land use includes activities such as interstate, state and county highways and roads; railroads; airports; and pipelines and utility transmission lines. Navigation is an important aspect of transportation; but as it does not involve an extensive use of land, it has not been included in this category.

The existing transportation system in the Great Lakes Basin has developed steadily, paralleling the population growth and economic development of the region. Different activities were more significant to the development of the transportation system at different times; but all have continued and are still operating in the Basin.

Surface runoff during the construction, operation and maintenance of highways, railroads, airports and pipeline and utility corridors carries pollutants from these facilities to nearby streams. These pollutants include: sediments, herbicides, deicing salts and abrasives, and exhaust emissions and other waste materials from vehicles and highway operations.

2.2 Scope of Study

Land use activities associated with transportation have been recognized as a non-point source of pollution for several years, but only recently have studies been undertaken to determine the extent to which the runoff from these activities affect the environment. Most of these studies have been directed at problems associated with pollution from the construction, operation and maintenance of highways, and very little information about the impact that other forms of transportation have on water quality is available.

2.3 Study Procedures

Information about water pollution caused by transportation activities was acquired in interviews with other university investigators, through interviews and correspondence with federal and state agencies, and by reviewing current literature and reports.

The major sources of the literature that were reviewed included professional journals, symposia proceedings and state and federal reports. Of particular value were the reports in the Environmental Protection

Technology Series published by the U.S. Environmental Protection Agency, and the Highway Research Record Reports of the Highway Research Board.

The facilities of the Northeast Water Resources Information Terminal (NEWRIT) for the Water Resources Scientific Information Center Retrieval System were used, but it could not return any abstracts related to the problem.

Section 3

REVIEW OF FINDINGS OF PREVIOUS STUDIES

3.1 General Description of Land Use Category

Transportation as a land use category within the Great Lakes Basin encompasses several different activities. Interstate, state and county highways and roads are the most significant transportation activity associated with water quality in the region; but surface runoff from railroads, airports and corridors for pipelines and utility lines also contributes to the pollutional load.

The impact that surface runoff from transportation activities within municipalities have on water quality is beyond the scope of this assessment, but has been considered in other reports in this series. Navigation as a transportation system has not been included.

3.2 Length of Time Activities in Practice

The development of the transportation systems that now exist in the Great Lakes Basin generally paralleled the population growth and economic development of the region. The growth of the various activities in the system were, however, greatly influenced by the major technological advances that occurred in the transportation field.

3.2.1 Highway Systems

It was not until the development of the automobile at the beginning of the twentieth century that surface runoff from highways began to have a serious impact on water quality. The demand for safer, paved road surfaces with longer sight-distances, reduced grades and improved drainage resulted in increased dislocations of the topography and accelerated erosion of the soil. The discharge of waste products from automobiles and from their exhaust systems, the control of vegetation along roadsides by herbicides and the desire for highway surfaces that are free of ice and snow increase the potential for pollution by chemical agents.

3.2.2 Railway Systems

Railway systems and canal systems for the transportation for goods and materials began in the region in the latter part of the nineteenth

century. Canals have all but disappeared from the Basin, and the construction of new railway facilities has almost been halted. However, surface runoff from rail facilities that are in operation still carries some pollution to nearby water courses.

3.2.3 Other Systems

Airport development in the Basin began around 1930, but more recent technological advances in the field of aviation, permitting large payloads and all-weather operation, have gradually increased the potential for pollution from these facilities.

The transmission of electrical power and the movement of material through pipelines lead to the development of utility corridors throughout the region beginning around 1920. Site clearing operations and the construction of these facilities cause some impact on the water quality in the Basin, but pollution loads are reduced as ground cover within the corridor is re-established.

3.3 Types and Nature of Pollutants

Some pollutants from transportation facilities, such as sediments and herbicides, are common to all the activities in the transportation category; while others, such as oils, heavy metals, and deicing salts are more specifically related to the operations involved with only one activity and not with the activity itself. For this reason, the following pollutants are described in two ways; 1) as pollutants common to all activities, and 2) as pollutants from one operation.

3.3.1 Pollutants Common to All Activities

3.3.1.1 Sediments

Construction associated with transportation activities is generally preceeded by the removal of all vegetative cover and top soil from the site. If required, the exposed sub-surface soils are then cut, and the excavated material is used as fill and compacted. Drainage channels are also cut and shaped as required. As a result of these activities, surface runoff from the construction site can carry large quantities of eroded soil to nearby water courses. The sediments are themselves relatively free of organic material and other nutrients, but they can cause siltation in streams unless properly controlled.

With the re-establishment of ground cover after construction is completed, the rate of erosion drops significantly; but maintenance operations, particularly those involved with the removal of vegetation from drainage channels, may temporarily increase the rate again.

3.3.1.2 Herbicides

Weeds, grasses and brush growing along highways and railroads, in utility corridors and at airfields are frequently controlled by the application of herbicides. This control is practiced for aesthetic reasons, for the management of the wildlife habitat, for improved access to the site and for fire prevention and control. As the herbicides are often applied on slopes, surface runoff tends to wash off the herbicide and concentrate it in the drainage system.

3.3.2 Pollutants Related to Operations

3.3.2.1 Salts

For safety reasons, most State and County highway departments in the region have adopted a "bare pavement" policy with regards to ice and snow removal. Sodium chloride and calcium chloride are the two salts most commonly used for this purpose. Airports also use deicing salts and other compounds to remove ice from active runways.

Other compounds are frequently added to deicing salts to help eliminate operational problems. Sodium ferrocyanide or ferric ferrocyanide are commonly used to minimize the caking of stored salt, and chromate and nutritious phosphate additives are used as corrosion inhibitors. The ferrocyanides are soluble in water and are capable of generating cyanide in the presence of sunlight. Chromium, like cyanide, is a highly toxic ion.

As all of the salts used as deicers are soluble in water, groundwater contamination may also result from excessive applications or improper storage.

3.3.2.2 Motor Vehicle Wastes

Various classes of waste materials are generated by motor vehicles operating on highway surfaces. Some of the more significant pollutants include fuels, lubricants and other fluids that leak from vehicles to the highway surface; fine particles worn from tires and from the pavement of the roadway; particulates from the combustion of fuel that are deposited from the vehicle exhaust systems; and miscellaneous wastes that include dirt, rust, etc., from the under carriage of vehicles.

The nature of these pollutants is not well defined, but probably include toxic metals such as lead, zinc and nickel from the fuels, lubricants and tires; tungsten from tire studs; asbestos from brake linings; and inert and reactive particulates from the other sources.

Additional pollutants associated with highways include litter along the roadside, dead animals, atmospheric "fallout" of air pollutants not generated by vehicular traffic, and all types of materials that are transported on or in vehicles that are accidently spilled on the road surface. These spilled materials may include relatively large amounts of toxic chemicals, inflammable substances and radioactive materials.

Many of these same products are also generated at air fields and along railroad rights-of-way.

3.4 State of the Art in Assessing and Quantifying Problems

Transportation as a land use category has long been recognized as one of the non-point sources of pollution; but, until the National Environmental Policy Act was passed in 1969, little documentation on the magnitude of the problems involved was available (5). The Act requires the preparation of impact statements for all federal projects that may affect the environment, and the need for information for these reports resulted in research and demonstration projects directed at assessing pollution associated with transportation activities.

The most significant transportation activity that the federal government has engaged in is highway construction, and the majority of the studies on the environmental impact from land uses associated with transportation have been performed in this area. The nature and source of many of the pollutants from other transportation activities, such as railroads, airports and utility lines, are very similar to those from highway construction and operation; and the work on highway pollution has indirectly provided information for these other subject areas. However, the rates of production and the amounts of material produced are different for the different activities, and additional work must be done before a complete assessment of the pollutional problems from the whole transportation category can be made.

3.4.1 Sediments

Sediments from soil erosion is conceded by most authorities to be the largest single stream pollutant (9). Physical damages that result from sediments include the siltation of reservoirs and lakes; the filling of harbors, and navigation channels; the alteration of the morphology and the stability of streams; and the clogging of drainage facilities along highways. Sediment also restricts the recreational use of water, interferes with water supplies, and disrupts stream and lake ecology.

It is estimated that the 6.44 million kilometers (four million miles) of roads in the United States contribute 50.8 million metric tons (56 million tons) of sediment to streams annually (9). Although not the largest source of sediment on an absolute basis, highways do have a high rate of sediment production on a unit area basis relative to other sources, particularly during their construction phase. Studies by Diseker and Richardson (6) show that bare road cuts erode at rates as great as 672 metric tons per hectare per year (300 tons per acre per year), and Wolman (23) states that the tonnage of sediment derived from erosion from an acre of ground under construction in highways and developments may exceed 20,000 to 40,000 times the amount eroded from farms and woodlands in an equivalent period of time.

Younkin (24) has reported sediment flow rates as great as 24.5 metric tons per hour (27 tons per hour) from the construction site of a new interstate highway, but he also reports that few data are available that can establish the contribution that highway construction sites make to the sediment problem. He has developed a regression equation that can be used to predict if a particular highway construction site would be a significant pollution source.

An inventory of 140,000 kilometers (87,000 miles) of highways in Wisconsin showed that there were 21,000 sites at which erosion was occurring. Nearly 73 percent of the sites were along town roads, 24 percent were along country roads, and only 3 percent were on state highways (4). No estimates of the amounts of sediments produced from these sites were presented; and in general, very little information about sediment production from established highways is available.

Many investigators have evaluated the different techniques that are used for sediment control at highway construction sites (3, 11); and several others have made attempts to modify an existing empirical erosion equation that is now widely used on agricultural lands for use at highway sites.

3.4.2 Pesticides and Herbicides

Numerous research projects have been conducted in the past and are currently funded to determine the state of current lake water and sediment contamination with pesticides. Probably even more numerous are the research projects associated with determining the toxic effects on fish and other aquatic life resulting from stream and lake pesticide contamination. Some research has been conducted in the past and is currently underway to determine the contribution of agricultural, municipal, and industrial operations to pesticide residues carried or deposited in river systems. Municipal investigations are generally associated with sewage treatment and disposal. Most studies in all categories have centered around the presence and influence of the common organochlorine (chlorinated hydrocarbon) insecticides such as DDT, dieldrin, etc. Consequently, although the literature, including that pertaining to the Great Lakes, is quite extensive on the concentration levels of pesticides in drainage, river, and lake waters and the effects upon organisms that live therein, very little literature is available on determining the actual source of contamination.

Herbicides have been and likely will continue to be used quite extensively in the control of unwanted vegetation along highway rights-of-way. Weeds and grasses growing in pavement seams and cracks are a potential contributor to highway and/or street deterioration. Unsightly weeds and brush along transportation right-of-ways require control and correction both because of the aesthetic values involved, the prevention of excessive weed spread to adjacent properties, and the management of wildlife habitat. The very nature of a highway system, with its broad asphalt or concrete surface plus the sloping berm and roadside right-of-way, concentrates the drainage system in such manner to promote potential runoff of herbicide residues. The evaluation of this possible source is vitally important in determining contributions to the water quality of the Great Lakes. It is a field of research that has been seriously neglected in the past and consequently should receive the necessary attention in the future.

A variety of herbicides are used throughout the Great Lakes Basin to control undesirable vegetation along state highway rights-of-way. The compound used most frequently to control broadleaf weeds is 2,4-dichlorophenoxyacetic acid (2, 4-D); while 2, 4, 5-tricholophenoxyacetic acid (2, 4, 5-T), used along or in combination with other herbicides, is frequently used for brush control. Other herbicides, such as amino-triazole. atrazine, bromacil, dalapon, dicambra, diuron, picloram, silvex (2-(2,4,5-T)P), and simazine are used to control more resistant vegetation or are used for spot applications at special locations along the highway system.

Many of the States in the Great Lakes Basin have prepared "herbicide manuals" that provide information concerning the types of herbicides available, their effectiveness against various types of vegetation and give recommended procedures for their application. The herbicides used in the region have been approved for use at labeled rates by State regulatory agencies or the U.S. Environmental Protection Agency.

Table 1

ANNUAL HERBICIDE USAGE BY SELECTED STATES

IN THE GREAT LAKES BASIN

STATE	HERBICIDE	USAGE	REMARKS
Ohio	2, 4-D	9,200 gal.	Used at labeled rates
	2-(2, 4, 5-TP)	500 lb.	along state's 4,700 mile
	Amizine	1,000 lb.	highway system in
	Dalapon	660 lb.	Great Lakes Basin.
	Dichlobenil	600 lb.	
	Others	700 lb.	
Pennsylvania	2, 4-D	12,000 gal.	Used at labeled rates
	2, 4-D & 2, 4, 5-T	15,000 gal.	along state's 44,600
	Picloram & 2, 4-D	3,000 gal.	mile highway system.
	2, 4, 5-T	4,000 gal.	
	Diuron	15,000 lb.	
	Bromacil	2,000 lb.	
	Simazine-Amitrol	15,000 lb.	
	Simazine	10,000 lb.	
	Others	600 gal.	
Indiana	2, 4-D	36,000 gal.	Used at labeled rates
	2, 4, 5-T	None	throughout the state.
	Others also used	--	

To convert gallons to liters, multiply by 3.785,
pounds to kilograms, multiply by 0.453,
miles to kilometers, multiply by 1.609.

Some of the approximate annual quantities of herbicides that have been used by selected States along highway rights-of-way in the region are shown in Table 1.

3.4.3 Deicing Salts

In the interest of highway safety, most States in the Great Lakes Basin have adopted a "bare pavement" policy with regards to ice and snow removal during the winter. A recent report of the Highway Research Board of the National Academy of Sciences stated that major highways in 33 snow-belt states would have to be temporarily closed or traveled under extremely hazardous conditions during winter storms if the use of deicing salts were to be discontinued (11).

The two salts most commonly used, sodium chloride and calcium chloride, come very close to being perfectly suited for snow and ice control. However, there has been much discussion about the impact of deicing salts on water runoff, soil bordering the highway, and plant and animal life subjected to the salt. Sodium chloride is generally used at temperatures above -12°C (10°F), for at lower temperatures the amount of salt required becomes prohibitively large. Calcium chloride is generally limited to temperatures above -34°C (-30°F). The cost of calcium chloride is approximately two to three times that of sodium chloride and is generally used sparingly. However, due to its rapid rate of solution, less calcium chloride is wasted during frequent plowing operations. When calcium chloride and sodium chloride are combined, the deliquescent calcium chloride supplies the triggering action to provide rapid melting action at all temperatures. Calcium chloride's speed of action produces a brine that sets off and sustains the melting action of sodium chloride over a longer period (14).

Most of the States in the Basin have issued directives for the use of deicing salts on their state highway systems in which they detail the roadway and weather conditions that trigger snow and ice removal operations. These directives also provide information about the types and amounts of chemicals that are to be used for different snow and ice conditions. In Ohio, for example, the following quantities of chemicals are used for two highway lanes:

Temperature	Road Conditions	Chemical Application
-7°C (20°F) or above and rising	Snow covered or Slushy	136 to 272 kg (300 to 600 lb) of salt per 1.61 km (mile).
-7°C (20°F) or below and falling	Dry Snow on dry pavement	Plow - no chemicals
-7°C (20°F) or below and falling	Pavement wet or freezing	136 to 272 kg (300 to 600 lb) of calcium-salt mixture per 1.61 km (mile)

Temperature	Road Conditions	Chemical Application
-12°C (10°F) or below	Packed snow or ice	Remove snow and apply abrasives. Apply calcium-salt mixture at above rates later.

The chemicals are applied to the middle one-third of the pavement. When conditions warrant it, calcium chloride is added to the sodium chloride for increased efficiency. For the 7,562 kilometers (4,700 miles) of State highways in the northern counties of Ohio that are in the Great Lakes Basin, over 162,000 metric tons (179,000 tons) of salt were used during the winter of 1973-74 for snow and ice removal.

Various substances are frequently added to sodium chloride and calcium chloride to prevent caking, inhibit corrosion and to mark the salt. It is unknown at this time whether these chemicals can create additional pollution problems. One additive is ferric ferrocyanide or commonly, Prussian blue ($Fe_4(Fe(CN)_6)_3$), which is added to salt to prevent caking. This appears to have no toxic effect (19).

Sodium ferrocyanide is also used as an anti-caking agent in salt. Sodium ferrocyanide's common name is Yellow Prussiate of Soda and has the chemical formula ($Na_4Fe(CN)_6 \cdot 10H_20$). It is quite soluble in water and will generate cyanide in the presence of sunlight (11). Another additive contains an agent that provides an electrical and physical barrier to corrosion. The material is mixed at the mine and contains sodium chromate, a hexavalent form of chromium. A concentration of 0.4% of sodium chromate in salt will result in a chromium to chloride ratio of 0.002; a hexavalent chromium concentration of 0.05 mg/l would be reached at a chloride content of 25 mg/l. A water supply influenced by runoff or seepage from a chromate treated salt would be adversely affected by the chromium long before the chloride became a problem (19).

Studies conducted by Hanes, et.al. (11), concluded the major rivers in the north-central and northeastern United States are not seriously affected by deicing salts applied to highways, and that it is the immediate roadside areas that are most seriously affected by the salt. They presented data for the chloride content of major rivers draining that portion of the United States receiving the bulk of deicing salt.

It has been estimated that highway salts contribute 11% of the total input of waste chlorides entering Lake Erie annually. The chloride concentration in Lake Erie rose from 7 mg/l in 1910 to about 23 mg/l in 1964 (12).

The situation is different for small streams or creeks and for small lakes and ponds which receive a substantial amount of their total volume from highways or other areas contaminated by deicing salts. Studies (22) have shown that extremely high chloride levels of 1,510 to 2,730 mg/l were found in the Milwaukee, Menomonee and Kinnickinnic Rivers at Milwaukee on January 16, 1969. High chloride levels were found in Meadow Brook in Syracuse, New York. Samples contained chloride concentrations in the range of 200 to 1,000 mg/l but frequently exceeded a few thousand mg/l. Samples taken in December showed about 1,000 mg/l chlorides in the Meadow Brook watershed.

The delay or failure of seasonal mixing due to salinity induced stratification has been observed in Beaver Dam Lake at Cumberland, Wisconsin, in First Sister Lake near Ann Arbor, Michigan (22), and Irondequoit Bay at Rochester, New York (7). All three cases were attributed to salt runoff from nearby streets entering these lakes. Because the quantity of oxygen in the lower depths is directly affected by seasonal mixing, it has been postulated that the addition of significant amounts of salt could contribute to the biological process of aging in lakes (eutrophication) and could contribute to the growth of blue-green algae which causes algal blooms (1).

Evidence has been presented by Feick, et. al. (2) that shows that contamination of fresh water environments by deicing salts runoff can interfere with the ability of bottom sediments to bind mercury and other heavy toxic metals. Other reports (13) indicate that dissolved salt may have some beneficial effects. Dissolved salt can influence toxicity of other pollutants. Hanes estimates that if a salt of a toxic metal other than a chloride is run into a river containing sodium chloride it will become a mixture of the original salt and the chloride of the metal, which could alter the overall toxicity beneficially.

The dissolved salt which enters streams, rivers, ponds and lakes acts as a weak electrolyte, changing the electrical conductivity of the aquatic system. In concentrations greater than 1% (10,000mg/l), all freshwater species of bacteria, algae, invertebrates, fish, and higher order plants are placed in immediate jeopardy (2). Living organisms survive in a delicate balance related to salt, each in accordance with its genetic limitations in special adaptations.

An example of a change in this delicate balance can be found in a tributary of the Genesee River in New York. As a result of high sodium chloride concentrations, primarily due to highway deicing salt runoff, the stream was changed from a trout stream to a stream dominated by salt tolerant species. The stream contained between 1% and 4% sodium chloride (10,000 mg/l to 40,000 mg/l).

Studies by Pollock and Toler (18) have shown that chloride concentrations in groundwater aquifers at test sites near highways, and analyses of water from wells near heavily salted highways and salt storage sites, indicate that a significant part of the salt applied to roadways is entering the groundwater system. Little, if any, dilution and mixing with native groundwater can occur because of the nonturbulent flow that prevails in the aquifer.

From the data obtained by Pollock and Toler (18), more knowledgeable decisions can be made regarding the location of highways to avoid areas of greatest potential danger to water supplies and the environment; the design of highway drainage systems to divert deicing chemicals along the least hazardous route; the selection of optimum application rates of deicing chemicals to ensure adequate deicing with minimum deterioration of water supplies; and prediction of maximum or minimum effect on the environment under various conditions.

3.4.4 Other Pollutants

3.4.4.1 Heavy Metals

Although some of the sources of heavy metals in the surface runoff from highways and roads have been identified, the contribution of each source to the total amount of a particular metal cannot be quantified.

A few studies have attempted to identify and quantify the heavy metal content in the surface runoff from urban streets (17,20), but very little research has been done to assess the magnitude of the problem associated with highways and roads. A portion of the results of a modest sampling program conducted on city streets, rural roads, and highways and at an airfield in the San Francisco Bay Area are presented in Tables 2 and 3 (22). These results generally indicate that city streets have higher concentrations of lead, zinc and copper; while highways have the highest concentrations of cadmium; and highways and roads have the higher concentrations of nickel and strontium. The values for airports are quite similar to the values for road surfaces.

In terms of loadings, the highway surfaces have the highest values; probably reflecting the higher particulate loadings on these surfaces.

Table 2

COMPARISON OF STRENTHS (mg/kg) OF DIFFERENT

PAVED SURFACE PARTICULATES FOR CERTAIN HEAVY METALS

METALS	CITY STREET	RURAL ROAD	HIGHWAY	AIRPORT TAXIWAY AND RUNWAY
Cadmium	3.8	0	9	6
Chromium	209	215	185	125
Copper	120	39	40	18
Iron	24,000	23,000	21,000	21,000
Manganese	440	860	370	310
Nickel	34	105	105	85
Lead	2,000	65	490	110
Strontium	21	50	50	0
Zinc	400	70	190	75

Table 3

COMPARISON OF LOADINGS OF DIFFERENT TYPES
OF ROADWAYS FOR CERTAIN HEAVY METALS

METALS	POUNDS PER CURB MILE		
	CITY STREET	RURAL ROAD	HIGHWAY
Cadmium	0.0037	0	0.058
Chromium	0.231	0.34	1.20
Copper	0.129	0.06	0.26
Iron	24.4	36	136
Manganese	0.468	1.35	2.39
Nickel	0.040	0.16	0.68
Lead	1.66	0.10	3.17
Strontium	0.022	0.078	0.32
Zinc	0.409	0.11	1.24

To convert pounds/curb mile to kilograms/kilometer, multiply by 0.284.

3.4.4.2 Miscellaneous Pollutants

There are several miscellaneous pollutants from highway activities such as roadside litter, wastes from dead animals, nutrients, organic wastes, spills and oils that also have some impact on water quality. However, it is generally felt that these pollutants are not significant when compared to the values generated from other sources in other categories of land use; and, as a result, these have received very little attention. As it is very recently that any assessments of the impact of surface runoff from transportation activities on water quality have been attempted; it is reasonable to assume that the impact from these miscellaneous sources will be determined after some of the more obvious and significant pollutants are investigated and assessed.

3.5 Transport Mechanisms of Pollutants

Contaminants from the surfaces of transportation facilities are washed into nearby receiving waters with the surface runoff generated by precipitation. Several mechanisms operate to remove the pollutants from the surface. The impact of raindrops, the emulsifying action of tires on highways and airports, and sheet flow act to provide good mixing and a continuous replenishment of fresh water to help dissolve soluble pollutants; while they also help dislodge particulates from the surface and carry them off as suspended material (21).

The sheet flow accumulates in drainage facilities constructed along the rights-of-way where it is conveyed rapidly to nearby water courses. Some of the runoff may infiltrate in the soil surfaces to replenish the soil moisture; or it may percolate through the subsoils to reach the groundwaters.

3.6 Natural Removal Mechanisms

There are several natural mechanisms that act to reduce the potential pollution from land use activities related to transportation. Some of these mechanisms prevent possible pollutants from being removed from their source, while others act to remove pollutants between their source and their disposal in water courses.

Vegetation acts to reduce erosion from a soil surface by reducing the impact energy of precipitation, by acting as a soil binder, by reducing the energy of surface runoff and by increasing the infiltration capacity of the soil.

Many classes of soils can adsorp cations from a solution by an ion exchange process. Thus, sodium and calcium from highway deicing salts, several of the heavy metals and some of the herbicides are effectively adsorped on clay sediments; and are removed from the runoff.

3.7 Probable Changes in Land Use Activities in Time

Transportation activities within the Basin will continue to develop in relation to the population growth and economic development in the region. However, much of the expansion will be accomplished by improving and enlarging existing facilities, and not by creating new transportation systems.

Automobile traffic will continue to increase through the region, but the rate of new highway construction could remain fairly constant and might even decline from its recent rate. Improvements to the existing highway system will also continue near its current rate. Mass transit services within municipalities will increase, but this will have little impact on the growth of highways and roads in the region.

Rail and air traffic will also increase throughout the region; but the existing railway and airport systems will probably be improved and expanded, and few new airports will be constructed to meet these increased needs.

Transmission of electric power from proposed generating stations will require that additional lines be constructed either within the existing utility corridors or within new ones.

3.8 Projected Future Pollution From Activities

The limited amount of information that is now available makes it very difficult to provide a current quantitative assessment of the impact that transportation activities have on water quality. Obviously, any assessment of future impact will be very tenuous.

However, there are some indications of future trends that should be noticed. Many states have limited the use of herbicides, the U.S. Environmental Protection Agency has specified the use of low-lead fuels for new automobiles, and many states have adopted sediment control practices for all highway construction. All of these activities should act to reduce the current rate of pollution from transportation activities in the future.

3.9 Review of Institutional Arrangements to Control Pollution

Because most transportation facilities come under different public jurisdictions, State level management and control has been limited to certain forms of transport facilities, such as state highways and public utilities. Likewise, State regulations tend not to apply to interstate facilities, such as railroads or airports, which come under the Interstate Commerce Commission's perview, and are thus subject to Federal rules and regulations concerning their environmental impacts and responsibilities under the U.S. Environmental Protection Agency's statutes and procedures.

Throughout most of the eight Great Lakes Basin States, there have been few, if any, State level management proceedings to regulate non-point source aspects of highways, railroads, or utility lines, except as they may be affected by general regulations controlling pesticide usage, sedimentation, and possibly certain regulations limiting solid waste disposal activities (16). Mention should be made of the potential utility of environmental impact statements for new transport construction starts for addressing these issues. By utilizing EIS procedures, states can begin to require preventive measures limiting the drainage of these materials to surface and groundwaters.

In the states with sedimentation control procedures, such as Michigan, Ohio, and Pennsylvania, sedimentation resulting from the construction of transport systems, such as highways, railroads, airports, and utility lines, can be limited.

In fact, Michigan's plan specifically requires that transport land use activities utilize sedimentation control plans, recognizing the potential for these land use categories to generate significant sedimentation loads (15). In the five other states voluntary advisory channels are available to provide temporary impoundments at highway construction sites to prevent siltation of waterways. Yet, voluntary actions alone may not be sufficient to curb sedimentation. Likewise, the states with sedimentation control ordinances are focusing principally on new construction starts and will not affect already existing highway facilities which may, due to their design, result in significant yearly loading of sediments in nearby surface waters.

Another aspect of highway non-point loadings concerns runoffs of salts, sand, and oils. There is currently no strategy among the eight Great Lakes Basin States to control these forms of surface runoffs from road systems. Oil seepages, resulting from automobile usage can leach into the soils and affect either surrounding surface water areas, or slowly impact upon groundwater quality. Likewise, seasonal applications of salts and sands can leach off in the springtime and affect surface water quality. There are few substitutes for salting roads in the wintertime to prevent icing, but most states do attempt to optimize their salting practices. Minnesota, for example, has provided legislation aimed at decreasing the use of salts on road surfaces in order to reduce the adverse environmental impacts from these activities; and Michigan has developed policies for covering and storing salts to help minimize pollution of ground and surface waters.

Pesticide and herbicide applications applied to roadside areas to curb vegetation come under general pesticide application laws at the State level. Nutrient loadings from the application of fertilizers to roadside strips of vegetation is not seen as a significant problem because of the limited nature of the activity. Currently, there are no programs for curbing or regulating fertilizer applications.

The major forms of erosion and sedimentation from railroads occur in the construction of tracks. However, this is not a significant activity at present and has not been an important point of focus for State water quality control. In fact, at the present time, the major focus upon controlling non-point source loadings from railroad operations lies in controlling herbicide applications to prevent certain herbicides from being used near adjacent surface water areas (2). There are no State programs to control sedimentation or the runoffs of oils or other compounds from existing rail lines.

Substantially, the same picture exists concerning airports. Few, if any, State regulations address the impact of airports' non-point pollutional impacts upon water quality. It does not appear that in the near future the transportation land use sector will be receiving attention at the State level in addressing their non-point impacts on water quality outside of management and control proceedings applied to new construction starts which may be incorporated under sedimentation programs or programs regulating herbicide and pesticide applications.

Section 4

SUMMARY OF ONGOING AND PROPOSED RESEARCH

The impact from construction, operation, and maintenance of the highway system on water quality is the subject of current study by the Environmental Protection Agency (EPA), the Federal Highway Administration (FHWA), and several state departments of transportation.

Previous and on-going work have dealt with various segments of the interaction of highway runoff and water quality and have often been of a limited or exploratory nature. Such information as effects of sediments or deicing chemicals, results from highway construction, street runoff, urban freeway runoff, and other related subjects have been or are being investigated, to varying degrees.

A study to develop a method for predicting the increase in sediment loading in a stream resulting from highway construction is the subject of a Pennsylvania Highway Planning and Research study at Bucknell University. The study involves collection, interpretation, and synthesis of turbidity, rainfall, construction, and stream discharge data. These data will be used to develop a mathematical model of a specific system--the construction of Interstate Route 80 through White Deer Creek Basin.

Data collection began in early 1968 preceding construction of the highway and the collection continued through completion of the highway in late 1970. Data on sediment concentration, turbidity, chloride concentrations, and hydrologic conditions are being compiled and analyzed. Some results from this project are already available and the final report is expected in 1974 (24).

Additional studies on highway construction, erosion, and acid and iron pollution abatement are underway for the Pennsylvania Department of Transportation.

A study jointly conducted by the U.S. Geological Survey and the Massachusetts Department of Public Works which deals with the "Effects of Deicing Chemicals Upon Surface and Ground Water" has been underway since early 1971. The objective of the study is to isolate and study the mechanics and transport of highway deicing chemicals by surface and ground water in relation to geologic, hydrologic, and highway maintenance conditions. Eight stream sites (surface water) and seven ground water sites have been installed and instrumented to monitor chloride content and movement.

A research study by the California Division of Highways on the influence of highways on water quality commenced during Fiscal Year 1973.

This study will establish and evaluate procedures for measurement, analysis, and mitigation of water quality degradation resulting from the highway construction process. Construction projects will be selected to monitor long-term effects of remedial measures in order to evaluate guidelines established in the planning and design phases. Studies on the "Influence of Highway Erosion Sediments and Waterborne Materials from Roadway Surface and Aquatic Biota" and "Water Pollution Aspects of Particles Which Collect on Highway Surfaces" have just begun by the California Division of Highways.

A comprehensive evaluation of the effects of highway maintenance and operation on water quality is currently being planned by the Federal Highway Administration. Earlier studies have dealt with parts of the problem but their scopes were limited and a larger perspective is desired. All aspects, from location selection for a roadway to the ramification of its operation and maintenance, shall be considered. Background data on water quality and ecological settings of basic types of highway sites and areas within the country shall be established. Changes of water quality with seasonal and storm intensity variations during runoff shall also be observed.

Section 5

AVAILABILITY OF TECHNOLOGY TO CONTROL POLLUTION

5.1 Nature of Existing Technology

There are several techniques that can be used to control pollution from non-point sources, and some of these have been applied very successfully to those land use activities that are associated with the transportation sector.

One technique is the modification of the source of the pollutant by either eliminating the contaminant from the source, or by substituting a non-polluting material for it. This type of modification has been attempted recently with the development of lead free and low-leaded gasolines, and the development of systems to control exhaust emissions from internal combusion engines. Among other benefits, it is anticipated that these developments will help reduce heavy metals and other pollutants in highway runoffs. The substitution of biological pesticides and herbicides for chlorinated hydrocarbons is another example of an attempt to help reduce pollution by this type of technique.

Another method used to control pollution from non-point sources involves binding the potential pollutant to its source so that it cannot be discharged with the surface runoff. Erosion control by soil stabilization, mulches and vegetation to limit sediment movement are examples of the application of this technique.

Treatment processes are not used too frequently to control pollution from non-point sources, but erosion control structures used during site clearing and construction activities at highway sites have proven very effective.

5.2 Management Practices

Most of the management practices used to control pollution from transportation activities have been described in other sections of this report. These types of controls are generally directed at limiting the amount or type of materials used during the construction, operation, or maintenance of transportation facilities. Existing State management schemes are limited to herbicide and pesticide control, and sedimentation control, but all States in the Basin attempt to optimize their use of deicing chemicals to limit the potential environmental problems from this activity.

5.3 Need for New Technologies for Pollution Control

Some of the current pollutants from transportation facilities do not lend themselves to existing removal or control technologies. Contaminants from accidental spills, deicing salts, roadside litter, and fluids that leak from vehicular traffic are not the types of materials that can be eliminated by existing treatment technologies.

However, there is a need for new technologies specifically aimed at removing contaminants such as trace metals and particulate materials from road surfaces before they are washed into adjacent water courses, or for separating suspended contaminants from storm waters before they reach the receiving stream. Highway drainage systems could be re-designed to concentrate surface runoff and convey it to single discharge locations for treatment, reuse or release under controlled conditions.

Containers used for the shipment of toxic chemicals and radioactive materials could be improved so that the contents would not leak or spill when accidents occurred. Contingency plans to limit the adverse effects of spilled materials have been adopted by regulatory agencies in many states, but more positive action is required.

The potential for new technologies to control unwanted vegetation by biological means should be developed, and the substitution of other methods of snow and ice removal should be explored.

Section 6

NEED FOR ADDITIONAL RESEARCH

Much of the research that has been directed at problems associated with pollution caused by transportation activities has only been completed during the past five years; and most of this work has been limited to areas related to highway construction, operation and maintenance. Several of these studies have dealt with the interactions between highway runoff and water quality, and have often been of a limited or exploratory nature. In addition, many of the more significant efforts have been directed at runoff from urban streets, and little work has been directed at highways and county roads.

Thus, there is simply not enough information yet available to assess the impact that the transportation sector has on the quality of the water resources in the Great Lakes Basin. Much more work has to be done to demonstrate and quantify the magnitude of the problems that exist. In particular, studies on the rates of production of the more significant pollutants must be performed; projects to quantify the amounts of waste materials that are produced must be completed; and more work to determine the impact that surface runoff from railroads, airports and utility lines have on the environment must be done.

Section 7

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Category A4 - Extractive Areas

EXTRACTIVE AREAS

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To be used as portion of the U. S. Task A Report on
GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by
the International Joint Commission - prepared in
partial fulfillment of U. S. Environmental Protection
Agency Contract No. 68-01-1598

July 1974

Section 1

SUMMARY

1.1 Introduction

The land-use category discussed in this report includes pits and open cuts, mainly for extraction of sand and gravel but also for clay and peat; quarries for removal of bedrock as crushed stone; open-cut mines for extraction of iron ore; underground mines for extraction of copper ore, gypsum, and rock salt; well fields, where brines are pumped for their content of dissolved salts; and oil fields, where brine is produced as an unwanted byproduct of oil production. Sand and gravel are produced in all eight of the Great Lakes states, crushed stone and peat in seven, and clay in five. Gypsum, salt, and brines are produced in Michigan, Ohio, and New York. Iron ore is mined chiefly in Minnesota and copper ore exclusively in Michigan.

Pollutants from these activities include suspended solids, or sediment, which may enter surface waters, and dissolved salts, which may enter both surface and ground waters. Of the two, dissolved salts appear to be the more serious problem.

1.2 Summary

Sand and gravel are recovered at more than 1,000 pits, and crushed stone at over 170 quarries, distributed throughout the Great Lakes region. Although some pollution of surface waters undoubtedly occurs, we have found no reference to the gravel or stone industries as polluters in any of the eight states.

Production of salt from mines, and of oil and salines from wells, in Michigan, Ohio, and New York has resulted in local contamination by chlorides. The most serious and long-range effects are found in ground waters. In general, sources of pollution of this type are known, and state regulations are either in effect or pending.

1.3 Recommendations

We recommend a preliminary survey of the gravel and stone industries, aimed at determining their seriousness as pollutants. Perhaps a random "spot check" of a few score of pits and quarries would indicate whether a wide-ranging system of monitoring ought to be established.

We also recommend continuing study of the problem of how to dispose of unwanted brines. This problem, serious now, will become more so in the future, when the production of salt and salt-derived chemicals will increase greatly. All possible methods of disposal should be evaluated.

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Section 2

SUPPORTING MATERIAL

2.1 Introduction

In this report we discuss the effect on the quality of surface and underground waters of the extraction of earth materials from pits, quarries, mines, and well fields. The materials discussed include seven "industrial minerals," namely sand and gravel, stone, clay, peat, gypsum, rock salt, and brines; and the ores of two metals, iron and copper. Although the production of these materials in the Great Lakes region is very large, both by tonnage and by value, of more significance to this study are the number and distribution of extractive sites. As shown in Table 1, sand, gravel, and stone are produced commercially at more than 1,200 localities. No single deposit accounts for more than a fraction of one percent of the total. Clay and peat are similarly dispersed, though on a far smaller scale. Gypsum, rock salt, brine, and the two metals are more limited in geological occurrence and therefore in number of extractive operations.

2.2 Scope of the Study

Information for the report was obtained from the state geological surveys of Minnesota, Wisconsin, Michigan, Indiana, Ohio, and New York. For Michigan -- the only state entirely within the Great Lakes Basin -- this contact was exceptionally valuable, because the Geological Survey Division of the Natural Resources Department is the administering agency of the state's Mine Reclamation Act and Mineral Well Act. We also received publications, letters, or telephone assistance from the Ohio EPA, the Water Resources Division of the U.S. Geological Survey, the New York State Department of Environmental Conservation, the Genesee - Finger Lakes Regional Planning Board, the Minnesota Pollution Control Agency, and the Robert S. Kerr Water Research Center of the U.S. EPA. A prime source of information on the nature and distribution of the mineral industry was the Minerals Yearbook (U.S. Bureau of Mines, 1972).

2.3 Study Procedure

The work was divided among the five members of the study group, with coordination and report-writing being handled by the principal investigator. Contacts with personnel of state and Federal organizations were made by phone, and a number of the persons called then furnished us with pertinent reports, both published and unpublished. In general, literature sources reviewed were those already known to the study group or suggested to them by those interviewed. The assumption was made that, for a rapid overview such as the present report, such procedure would be adequate.

Section 3

REVIEW OF FINDINGS OF PREVIOUS STUDIES

3.1 General Description of Land-Use Category

3.1.1 Extractive Industries Other Than Salt and Brines

In terms of tonnage, gross value, and number of extractive operations, by far the most important mineral resource of the Great Lakes Basin is sand and gravel. These materials are used solely as aggregate in concrete. Because they command only a low price per ton, sand and gravel will not bear high transportation costs and must be produced close to their point of use. Thus there are hundreds of pits, scattered throughout the region, each supplying a local market (Table 1).

A close second in gross tonnage is crushed stone. In the Great Lakes region, nearly all this stone is limestone or its close relative, dolomite. Much is used as aggregate in concrete, but large tonnages also go into the manufacture of cement, lime, refractories, and steel. The clay that occurs in the region is common clay; no high-quality china clays or other premium varieties are known. Between 80 and 90 percent of the clay produced in Michigan and in the northward-draining counties of Ohio -- the major producing states -- went into the manufacture of portland cement, and was thus captive production of cement companies. Most of the remainder was used for ordinary brick and tile, and for expanded or lightweight aggregate. Peat -- nominally a fuel, but used in this country as a soil conditioner and potting medium -- is produced in small operations in seven of the eight Great Lakes states.

Gypsum is produced at two mines and two quarries in Michigan, the nation's leading state in gypsum production; at one mine and one quarry in Ohio; and at three mines in New York. Wallboard-manufacturing plants are located adjacent to most of the mines and quarries. The gypsum is taken directly to the plant, where it is crushed, ground, and calcined. A little gypsum is used as a retarder in portland cement.

Surface mining for iron has long taken place in Minnesota (Vermilion and Mesabi ranges), Michigan (Gogebic), and Michigan (Menominee, Marquette). The Mesabi Range today produces about 75 percent of U.S. iron ore, the other ranges being relatively inactive. Copper mining is confined to one underground mine on the Upper Peninsula of Michigan.

TABLE 1 - EXTRACTIVE INDUSTRIES OF THE GREAT LAKES BASIN, 1970[1/]
(Exclusive of Oil and Gas)

Commodity	MINN.	WIS.	ILL.	MICH.	IND.	OHIO	PA.	N.Y.	Total
Sand-and-gravel pits	61	230	13	453	57+	109	1	113+	1038+
Crushed-stone quarries...	1	88	3	27c[2/]	9c	27		16c	171+
Clay pits....		2	2	16	2	5			27
Peat bogs....	2c	1c	2c	11c		4c	1	3c	24+
Gypsum mines or quarries				4		2		3	9
Salt mines...				1		2		2	5
Brine fields.				6c		2c		3c	11+
Iron mines (open-pit).	1c								1c
Copper mine..				1					1

[1/]Data are only for those counties in each state that are wholly or partly in the Great Lakes Basin.

[2/]For some states, only the number of counties can be tabulated. Numbers designating counties are followed by the letter c.

Source: Minerals Yearbook 1970 (U.S. Bur. Mines, 1972).

3.1.2 Salt and Brines

Thick beds of rock salt underlie the southern peninsula of Michigan, the northeastern counties of Ohio, much of western Pennsylvania, and New York south of a line approximately from Syracuse via Rochester to the southwestern corner of the state. These salt beds constitute a valuable mineral resource, which is mined in underground workings at one locality in Michigan, two in Ohio, and two in New York. The salt produced is used for ice control on streets and highways, and in the chemical and food industries.

As geologic time has passed, waters have come into underground contact with the salt beds and have dissolved salt, forming brines. These brines are valuable mineral resources. Naturally occurring brines are pumped to the surface in an area of central Michigan centering around Midland, and in Mason and Manistee counties on the western edge of the state. At these places the brines yield not only sodium and chlorine, the constituents of common salt, but also numerous other elements, such as bromine, iodine, magnesium. At other localities, notably Painesville in northeastern Ohio, and at the south end of Seneca Lake, New York, wells are drilled to the rock salt, water is pumped down to dissolve it, and the resulting "artificial brine" is brought to the surface. Here it is evaporated to yield salt, or it may be pumped directly into a manufacturing complex, where it forms the raw material for production of sodium chemicals and chlorine chemicals.

Much brine is also brought to the surface as an unwanted byproduct in the production of oil, in the oil fields of Michigan, Ohio, and New York. These brines have been locally sprayed on rural roads for dust control, but most of them constitute a waste that must be disposed of.

3.2 Historical Development of Extractive Industries

Large-scale production of sand and gravel began after the introduction of steel-frame construction in 1885, which brought with it a demand for concrete. Extraction of crushed stone started somewhat later, after development of such equipment as crushing machinery, the steam drill, and dependable explosives. Tremendous expansion of the aggregates industry has taken place since World War II, as a result of the boom in housing, commercial and industrial building, construction of highways and airports, and similar uses of concrete.

By contrast, clay, gypsum, and salt have uses that go far back into antiquity, and all have been extracted since shortly after the first white men came into the Great Lakes region. Clay yielded brick and tile, and gypsum gave plaster and mortar, for the early builders. The Onondaga salt springs, near Syracuse, New York, were known in 1788, and soon thereafter salt was being produced by evaporating the brine in kettles over fires. Wells were later dug for the brine. By the mid-1800s, 140,960,000 litres (4 million bushels) of salt was being produced annually. Natural brines at Saginaw, Michigan, were tapped in 1860; they were especially well situated for the meat-packing industry of Chicago. Production of salt and brines in Ohio has a similarly long history.

From the mid-1800s to about 1953, the Lake Superior region produced the bulk of the iron ore used in the nation's steel industry. In 1953, the peak year, production reached 97 million metric tons; by 1968, it was 57.5 million. The decline can be traced to three factors: depletion of high-grade reserves, the "direct-shipping" ores; imports of foreign ores; and the technological revolution that allowed production of highly desirable concentrates from previously unusable low-grade ores (taconite). Future production will be of taconite and similar ores on a very large scale, using production-line techniques. It is expected that about one-half of total U.S. needs for iron ore will be met by Lake Superior production; of this amount, Minnesota will produce 75 to 80 percent.

By 1880, Upper Michigan was the nation's leading producer of copper. Output declined steadily through 1954, except for temporary increases in the late 1920s and World War II. A new mine in Ontonagon County doubled the state's copper output, and by 1962 it was 74,000 tons, or 6.0 percent of national production. Activity at this mine remains strong, but the future of the world copper market is beset with uncertainty.

The history of iron and copper mining in the Lake Superior region is summarized by Deshpande and Dworsky (1973, p. 2-6 - 2-13).

3.3 Pollution Resulting from the Extractive Industries

3.3.1 Nonmetallic Extractive Industries Other Than Salt and Brines

In the average sand and gravel operation, pit-run material is washed, screened, and stockpiled in several sizes, from pebbles 3 inches or more in diameter down to fine sand. The effluent from washing may contain unwanted sand (grains larger than 1/16 mm in diameter) or silt (1/16-1/256 mm), but is seldom burdened with clay (finer than 1/256 mm). Little clay occurs with sand and gravel, because the water currents that originally deposited these materials -- meltwater from the ice sheets of the Pleistocene age -- were too swift to allow clay-size material to settle out. The effluent can be readily cleared of its sand or silt by settling in a pond, and this is commonly done, so that the water can be used over again. (People in the sand and gravel business say, "We use the water till we wear it out.") Such waters carry little dissolved material, other than what they originally had, as the constituents of sand and gravel are by their very nature resistant to chemical attack and thus unlikely to yield a solution load to waters that come into contact with them. Nearly all the waters used at sand and gravel pits are surface waters. At some deposits, the desired material lies below the water table, and is extracted from under water by dragline or clamshell bucket. The bodies of water that are left in worked-out pits constitute park lakes, waterways for residential development, or ground-water recharge basins. Pre-planning toward these goals is standard procedure at many of the larger pits. Constraints on the industry are imposed chiefly by (1) competition with urban expansion for close-in sites of production; (2) community objections to noise, traffic, and unsightliness; and (3) necessity for reclamation of mined-out areas.

The following paragraphs are quoted from a letter of March 14, 1974, from E. K. Davison, Director of Environmental Affairs of the National Sand and Gravel Association. In cooperation with a man from the EPA, Mr. Davison "solicited information on discharges from a number of our member companies around the country."

> I found in my limited survey that in about one-fourth of those operations which discharge into a public watercourse the discharge is simply a pump-down to keep a pit dry or, more usually, to evacuate excess water, rather than a discharge of wash-water from the screening plant. Such pumps appear to be well removed from portions of pits used for receiving screening plant washwater.
>
> The few analyses of pump-down discharges that I received indicated that suspended solids contents exceeding about 12 to 15 mg/l would be unusual. Analyses furnished me for discharges involving settling ponds for process water (as distinguished from pump-down discharges) showed a median suspended solids content of 139 mg/l. These were mostly "one-shot" samples taken for filling out the permit applications under the Refuse Act Program. One of the EPA Regional Offices in a study of both sand and gravel and crushed stone operations found a median of 55 mg/l for suspended solids. This particular report corroborated the observation of both the EPA Effluent Guidelines man and this Association office that volumes of discharges related to tons of production vary so widely that only the pollutant concentration method, as distinguished from loading per unit of production, is practical. Suspended solids and pH are the usual parameters of concern. The pH is no problem in discharges from sand and gravel operations.
>
> I have never heard of any concern or contamination of underground water provided the pits or ponds are receiving only process water from sand and gravel handling.

Pollution of waters by the sand and gravel industry may be appreciable, but if so it is largely unrecognized. In the sources consulted for this report, we have found hardly any mention of this subject. It seems to have given rise to little if any regulatory legislation.

Water may enter into the crushed-stone operation if the stone is naturally muddy or shaly, but this is an undesirable condition and at most quarries the quality of the stone is such that washing is not necessary. Water may be clarified by settling in ponds, but it is probable that some mud and silt goes into adjacent waters. At a limestone quarry on Kelleys Island, Ohio, for example, silty water was discharged for some years into Lake Erie, where it produced a little delta. Slight chemical effect is generally made on waters that come into contact with limestone or dolomite for a few minutes during washing.

Production of clay consists simply of extraction from a pit and removal to the plant. The industry is relatively small in the Great Lakes area and little effect on quality of waters is to be expected. Much the same is true of peat. This substance is dug up, shredded or otherwise dry-processed, packed, and shipped.

Between 16.3 and 19.7 million liters (4.3 and 5.2 million gallons) of water are pumped every day from a gypsum mine on the Port Clinton peninsula on the south side of Lake Erie (L. Purcell, U.S. Gypsum Co., personal communication). This exceptionally large quantity of water is explained by the presence of fractures through which lake water is entering the mine. (Variation in the above figures reflects water level in the lake, which in turn depends on prevailing winds.) In May, 1974, the water had a pH of 7.72, an average sulfate content of about 1,380 mg/l, and total dissolved solids averaging about 2,400 mg/l (R. Sturtz, Ohio EPA, personal communication). Some of the water is used in the sink-float process of upgrading the gypsum in the mill; the rest is discharged into Sandusky Bay, an arm of Lake Erie. The state EPA does not consider this discharge as a serious source of pollution, owing to the great dilution that it immediately undergoes. However, the Ohio EPA does not have standards that apply to lakes and bays, only to streams. It is not possible to generalize from one gypsum mine to others, as the amount and nature of water will vary with the local conditions. Water pumped from a gypsum mine in western New York, for an example, is pure enough to be diverted, by contract with the state, into a stream at a guaranteed rate of 5,677 liters (1500 gallons) per minute (R. Runvik, U.S. Gypsum Co., personal communication).

> Coal was at one time mined by underground methods in Bay, Genesee, Huron, Midland, Saginaw, and Tuscola counties [Michigan] from the Saginaw Formation of the Pennsylvanian System. Coal production ceased due to the thinness of the coal beds, the low grade and high sulphur content of the coal, and the high cost of underground mining. No future coal production is anticipated (U.S. Bureau of Mines, 1971, p. 5-127).

Effects, if any, of the discontinued mines on the quality of waters is not known. Deleterious effects of exploratory drill holes are discussed in Section 3.5.

3.3.2 Extraction of Salt and Brines

Unlike sand, gravel, and the other relatively inert substances discussed in the preceding section, salt is readily dissolved in the earth's waters, and the resulting brines are chemically active. When discharged on the ground surface, they sterilize the soil and kill vegetation; if discharged into a water supply, they give an undesirable salty taste to drinking water. Thus "chloride contamination" is a serious and continuing problem in brine fields, and also at salt mines where mine seepage must be disposed of. Waste brines have traditionally been flushed into streams or lakes. For example, brine has been discharged into the south end of Seneca Lake, New York, for nearly 100 years; in addition, operational leakages, spillages, and the former practice of stockpiling salt have undoubtedly contributed chlorides to the lake.

> Seneca Lake had an average chloride content of about 100 mg/l from about 1920, when first analyzed, to 1960. From 1960 to 1970, chlorides increased to about 200 mg/l. Since 1970 there has been a levelling off or slight decline in chloride. Salt-plant operations (possibly including leaky abandoned salt wells) are suspected causes (R. M. Waller, U.S. Geol. Survey, personal communication).

Newly established standards controlling stream pollution in New York, discussed in Section 3.9.3, will eliminate disposal of brines into streams in that state.

Waste brines may be pumped into ponds, in the hope that the water fraction will evaporate and the salts remain as precipitates. This practice, however, is successful only in arid or semi-arid regions, where evaporation exceeds rainfall (U.S. EPA Rept. EPA-430/9-73-011, p. 94). Under the Great Lakes climate, evaporation is slow and much of the brine infiltrates into the ground-water supply. Leakage and filtration through the dikes at evaporation ponds is also a hazard. A third method of disposal is to pump the brine back into the ground. It may be pumped into an oil-bearing zone, where it may help to repressure the reservoir and act as a water-flood to move more oil toward well bores; needless to say, this is a technique that requires considerable specialized knowledge. Or, the brine may be pumped into a deep porous and permeable bed or rock that will absorb and hold it. Much remains to be learned about this technique.

In sum, the production of salt and brines appears to be the most damaging extractive industry to the quality of the water supply in the Great Lakes region.

3.3.3 Mining of Iron and Copper Ores

Crushing, screening, washing, and gravity beneficiation are the chief processes that iron ore goes through at the mine. The wash water, much of which used to run off and produce "red water" in the streams draining the Mesabi and other ranges, is now clarified in settling ponds and is re-used. The development of closed-water systems in the tailings-disposal areas has been notably successful in reducing pollution (R. K. Hogberg, Minnesota Geological Survey, personal communication). The St. Louis River at Duluth shows the lowest longtime average annual sediment yield of any large river system in the Great Lakes Basin: 1 metric ton per km^2. (The Cuyahoga River south of Cleveland shows 88 metric tons per km^2.) The St. Louis River drains a large part of the Minnesota iron ranges. Planning subareas 1.1 and 1.2 of the GLBC, which contain the iron and copper mines, have the lowest gross erosion rate of all subareas in the Basin, as indicated in Table 2. The effects of mining are not mentioned in a study of suspended sediment in streams tributary to Lake Superior (Callahan, 1973).

3.4 Assessing and Quantifying the Problems

To assess the extent of pollution by the region's extractive industries other than salt and brines would involve visiting and inspecting 1,000 to 2,000 separate pits, quarries, and mines. Most of the pollution would be in the form of sediment in water. Quantifying the observations would involve measuring the discharge and determining the concentration of suspended solids at each occurrence. To be of value for purposes of comparison and totalling, such measurements should be taken at times of average or "normal" discharge and pollution. It is doubtful whether the results would be reliable and significant enough to warrant the effort and expense.

Determining a water's chloride content is a relatively simple process of titration and is done on a routine basis. It gives quantitative results that are comparable with a large body of similarly expressed data from waters at many localities. Standards of comparison are generally known and accepted. For example, the U.S. Public Health Service recommends that drinking water contain no more than 250 mg/l of chloride. Thus chloride contamination may be readily assessed and quantified.

TABLE 2 - GROSS EROSION RATE, SUBAREAS 1.1 AND 1.2

Subarea	Tons per Acre per Year Current	1980	2020
1.1	0.33	0.34	0.32
1.2	0.24	0.23	0.22

Source: Great Lakes Basin Framework Study, Appendix 18, Table 4, 1970.

3.5 Mobility of Pollutants

The mobility of sediment discharged into water bodies, as from washing plants at gravel or crushed-stone operations, depends on (1) the coarseness of the material, and (2) the kinetic energy, if any, of the water body. Sand and silt discharged into a pond or other standing water will settle promptly. Clay may remain in suspension for a long time. If the discharge is into a stream, natural sorting will take place: coarse sand will settle nearest the point of inflow, and silt farther away; clay will very likely not come to rest until the stream reaches a body of quiet water where energy is at a minimum and settling can take place. Sources of severe pollution by suspended solids should be relatively easy to spot because of the above-described method of dispersal.

Brines are highly mobile. Those brines that seep into streams or lakes may cause contamination, or they may undergo such great dilution by fresh waters that the stream or lake waters show only slight effects. The chloride concentration of streams during low flow throughout Michigan rarely exceeds 35 mg/l, even though brines are present in the shallow subsurface of much of the state.

Waters that remain underground can hardly purge themselves. If fresh waters become contaminated with chloride, they may remain contaminated indefinitely. Under natural conditions, underground waters generally move very slowly; but when man enters the picture he may upset the hydrostatic equilibrium, causing rapid migration of brines into fresh-water zones. From 1860 on, hundreds of wells were drilled in the Saginaw Valley of Michigan, in the search for coal and valuable brines. Few if any of these wells were lined with casing, with the result that they served as conduits through which highly mineralized brines from deep in the rocks rose and mixed with near-surface fresh waters. Deterioration of ground-water quality was noticed at Saginaw as early as 1906 (Bowman, 1906); it was attributed to abandoned brine wells. At Lowell, near Grand Rapids, Kent County, the chloride content in a municipal well increased from a trace in 1933 to 925 mg/l in 1941. The major cause was leakage of brine through abandoned oil and gas wells (Deutsch, 1963). A well drilled in Lansing in 1956 contained less than 100 mg/l, but after two months of operation the chloride content had risen to about 900 mg/l. The probable source of contamination was an abandoned brine well (4,575 mg/l of sodium chloride), which had been drilled in 1867. Smith (1944) has described the deleterious effects of brine migration on overlying fresh-water aquifers in Michigan, and Smith and Frye (1945) have reported on the history of brine contamination in the Saginaw Valley.

Many thousands of shallow shot-holes are drilled each year in Michigan by oil companies for geophysical testing; borings are made for foundation work; and holes are drilled by cement companies or others to test the limestone bedrock. All these holes are uncased, and many have acted as pipes through which mobile brines move upward. The result has been chloride contamination of water supplies that continues even today.

The gross areal extent of ground-water contamination in southern Michigan is not known. Probably most individual areas do not exceed a few acres. In districts where ground water is pumped, the resulting decrease in hydrostatic head might lead to more widespread contamination, through open holes and inter-aquifer leakage.

In considerable areas of Wood and Hancock Counties, northwestern Ohio, surface and ground waters have a higher-than-normal salinity. Near Portage, Wood County, the Portage River at periods of low flow commonly contains more than 150 mg/l of chloride. It seems very likely that contamination is the result of leakage of brines from the old Lima oil and gas field. This field was discovered in 1884, reached its peak of production in 1896, and is nearly exhausted today. According to Orton (1888), oil seeps into wells and excavations in the Findlay area antedated discovery by 50 years. No doubt deep brines have been moving upward for scores of years, in large part via old oil and gas wells which were never cased or in which the casing has long since rusted away.

Another situation is exemplified by the artificial-brine industry near Syracuse, New York. The wells are at Tully, 27.3 km (17 miles) to the south; the brine is pumped to a plant near the west shore of Onondaga Lake, where caustic soda and soda ash are produced. Lagoon effluent and leachate from stock piles contribute high chloride content to Ninemile Creek and Onondage Lake. The situation is discussed in two reports (Kantrowitz, 1970; Shampine, 1973).

3.6 Natural Renovative Mechanisms

Water that is contaminated only by suspended solids, as at a gravel or crushed-stone plant or taconite mine, undergoes renovation if these solids are allowed to settle out. Water thus clarified may be re-used in the plant or discharged into water bodies without deleterious effects.

There is essentially only one natural mechanism for renovation of chloride-contaminated waters. This is dilution by fresh waters. Unless the pollution load is very heavy and the water body is small, such dilution ordinarily reduces the chloride level to acceptable values. Where underground waters have been invaded by brines, there is no natural means of renovation.

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3.7 Probable Changes in Extractive Industries with Time

It will be seen from Table 3 that all the commodities discussed in this report are expected to increase markedly in production well into the next century. Salt, the major source of water pollution, was first produced for food seasoning and preservative, but the salt industry has long since become a mainstay of the chemical complex, and the bulk of today's production goes into the sodium and the chlorine families of basic chemicals. This being the case, there is little doubt that demand for salt will continue to rise. It is expected to more than quintuple by the year 2020.

Production of both iron and copper is expected to increase moderately in the coming decades. Deleterious effects on water quality may increase proportionately, although it seems entirely within reason to believe they might be held to present levels or even made to decrease, as more is learned and applied to control of tailings accumulations and the drainage from them.

3.8 Seriousness of Future Pollution Problems

For mineral resources other than salt and brines, there is little basis for prediction. A reasonable assumption seems to be that pollution will increase in roughly the same proportion as production increases. We believe the over-all effects will continue to be slight.

As for salt and brines, it is hard to avoid the conclusion that chloride contamination will increase markedly unless firm governmental controls on extractive and manufacturing processes are imposed. Hazards to be expected are exemplified at a New York site, where a company started mining rock salt about 2 years ago. While working on a deep-well disposal system, they have stored mine seepage and plant wastes in holding ponds. There has been leakage, dike failure, and production of a fine saline dust, all of which have caused problems to the water supply (R. M. Waller, U.S. Gel. Survey, personal communication). With five times as much salt being produced in 2020 as is being produced today, it is clear that drastic controls and monitoring systems will be necessary.

3.9 Review of Programs of Controlling Pollution

A comprehensive description of methods for controlling pollution from extractive operations of all types is given in two reports of the U.S. Environmental Protection Agency (1973).

TABLE 3 - PROJECTED MINERAL PRODUCTION, GREAT LAKES REGION,
BY SELECTED COMMODITIES
(Thousands of Metric Tons)

Commodity	1968[1/]	1980	2020
Sand and gravel...	116,981	155,276	464,876
Crushed stone.....	100,298	130,151	387,537
Iron ore.....	57,548	66,605	126,748
Salt.....	W[2/]	20,993	94,911
Clay.....	3,755	4,600	11,663
Gypsum.....	W	2,005	3,692
Peat.....	237	262	427
Copper ore.....	68	91	299

[1/] Actual.

[2/] Withheld to avoid disclosing individual company data.

Source: Great Lakes Basin Framework Study, Table 5-103 (1971).

3.9.1 Michigan

Michigan's Mine Reclamation Act (Act 92, 1970, amended by Act 123, 1972) provides for the reclamation of lands subjected to the mining of coal, gypsum, stone, and metallic ore (but not sand and gravel, clay, or peat). The effects of mining on water pollution is one of the objectives of a comprehensive survey mandated by the act. This survey is currently under way and is expected to result in promulgation of rules governing (among other things) pollution of public waters. Supervisor of the Act is the chief of the Geological Survey Division of the Department of Natural Resources. Completion of the comprehensive survey is hoped for in 1974.

The Water Resources Commission Act (Act 245, 1929, as amended, effective April 15, 1973) creates a Water Resources Commission "to protect and conserve the water resources of the state, to have control over the pollution of any waters of the state and the Great Lakes . . ." etc. Mining operators must obtain permits to discharge wastes into public waters.

Michigan's Mineral Well Act (Act 315, 1969) is intended "to provide control of the drilling, operating and abandoning of mineral wells to prevent surface and underground waste; . . . to provide for inspecting, repairing and plugging of mineral wells; . . ." etc. Supervisor of the Act is the chief of the Geological Survey Division. Full enforcement of this Act should go far toward controlling pollution from mineral wells.

Material in this and the following paragraph is taken from Deshpande and Dworsky (1973). Official Michigan policy states, "The business of mining and beneficiating low grade iron ore . . . is declared to be in the public interest and necessary to the public welfare, and the acquisition of private property for development of an adequate water supply, the necessary storage, processing and treatment of liquid and solid wastes . . . is declared to be for a public purpose." The Michigan Department of Natural Resources is authorized to acquire by condemnation land for settling ponds, storage basins and treatment, transportation and other facilities -- this land to be leased to the mine operator if he has already secured 75 percent of the necessary land (Michigan Statutes 425.171, P. A. 1968, no. 314).

"It is the policy of the State to develop and continue mining in the Upper Peninsula. It is in the public interest and for the public welfare." Because beneficiation needs water, the use of water in the processing of low grade iron ore is declared to be in the public interest (Michigan Statutes 323.251, sec. 1, P. A. 1959, no. 143).

3.9.2 Ohio

The surface waters of Ohio are protected under Regulation EP-1 of the Ohio Environmental Protection Agency, effective July 27, 1973, entitled "Water Quality Standards." Sources of water pollution are regulated by applying effluent standards and issuing permits, under the general style of the Federal Water Pollution Control Act, Section 402, "National Pollution Discharge Elimination System."

Regulations under Ohio's oil and gas law (Revised Code Chapter 1509) prohibit the contamination of surface or ground waters by disposal of oil-field brines, but drilling operations are so numerous and state regulatory agencies so undermanned that adequate control is not exercised.

Ohio has no other pollution-control program for its subsurface waters.

3.9.3 New York

New standards to control stream pollution became effective in New York on March 27, 1974. These are set forth in the State Department of Environmental Conservation's "Classifications and Standards Governing the Quality and Purity of Waters of New York State." Title 6, Official Compilation of Codes, Rules and Regulations, Part 702.1, refers to "Class A-Special (International Boundary Waters) (Great Lakes Water Quality Agreement of 1972)." Quality standards include, among other items, requirements that total dissolved solids shall not exceed 200 mg/l, and that suspended solids that will be deleterious for any best usage will not be allowed. Legislation to support these measures comes under Article 17 of the Environmental Conservation Law (formerly Article 12 of the Public Health Law) (W. E. Loveridge, NYSDEC, personal communication and enclosures).

It may be significant that a comprehensive survey by a regional planning agency, with full attention to major sources of pollution, especially of ground water, does not mention deleterious effects of any extractive industries. This report is listed in the references under Genesee-Finger Lakes Regional Planning Board.

3.9.4 Wisconsin

In Wisconsin, according to Wirth et al. (1973), "current state laws are adequate to control water pollution related to mineral resources." Deshpande and Dworsky (1973), on the other hand, find that the state lags behind Michigan and Minnesota in creating formal policies and safeguards against environmental damage that might be caused by mining. If mining operators are to divert waters for their use, they must secure a permit

from the Department of Natural Resources. "The Department shall impose such conditions in the permit as it finds are reasonably necessary in the public interest for the restoration of waters after completing of the mining operations or cancellation of the permit, for the orderly disposal of waste or tailings . . ." (Statute 107.05, sec. 9). Statute 144.01 subjects "unnecessary siltation" from quarries and gravel pits to regulation. Statute 29.29(3) prohibits discharge of deleterious substances into the state's waters. But all these regulations apply only if the mining operation requires diversion of water for its use. The regulations do not apply to ground water.

3.9.5 Minnesota

All state agencies in Minnesota are directed to cooperate with and assist the Pollution Control Agency, which was formed in 1967. The Division of Water, Soil and Minerals of the state's Department of Natural Resources is in charge of mining regulation. The commissioner of the Department may grant permits for drainage diversion and control or use of water where necessary for mining of iron ore, taconite, or copper -- if it is necessary and there is no other alternative, if it will not substantially impair the interests of the public, and if it will not endanger health or safety (Deshpande and Dworsky, 1973). If a permit is granted, the Commissioner will also prescribe the requirements for restoration of the waters to their former condition. This may require the mining operator to post a bond (Statute 105.64). Mine dewatering is monitored by the state's Conservation Department.

3.9.6 Other States

Only three Illinois counties drain into Lake Michigan. The state is currently limiting all discharges into the lake, and no new ones are to be allowed.

There are large sources of pollution in northern Indiana, but these are industrial plants rather than extractive industries. Urban runoff, industrial wastes, and waste-treatment processes are the significant sources of pollution of Lake Michigan and the streams tributary to it.

Only Crawford and Erie counties in Pennsylvania are tributary to Lake Erie. The state's "Clean Streams Act" and other legislation should be more than sufficient to control pollution from the single sand and gravel pit and the one peat bog reported to be active in this area.

Section 4

SUMMARY OF RESEARCH PROGRAMS

4.1 Nature of Studies

In the limited time available, it has not been possible to make a study of "current or proposed research, demonstration, or monitoring programs that may help answer basic questions." Such a study is desirable and should be made.

State-of-the-art studies of pollution produced by certain extractive industries are currently under way at the Robert S. Kerr Water Research Center of the U. S. Environmental Protection Agency, in Ada, Oklahoma. Reports are soon to be published on the significance of water pollution associated with the sand and gravel, uranium, and oil-shale mining industries. Research is also being done on sealants for mine-tailing sedimentation ponds. Studies bearing more directly on the extractive industries of concern in this review are in the discussion stage at the Center.

With a grant from the Federal Water Pollution Control Agency, researchers at the University of Minnesota and their associates have made a study of the role that peat might play in combating water pollution (U.S. Bur. Mines, 1972, p. 389).

The following news item appeared in the magazine Rock Products, July 1974.

> The Environmental Protection Agency finally is moving ahead with its previously announced study of mining effluents prior to establishing guidelines and new source performance standards. The consulting firm Versar, Inc., has been awarded the contract; it will meet with the trade associations to identify critical factors and review control methods currently in use. $3.5 million has been released to EPA by the Office of Management and Budget to participate in the joint U.S.-Canadian investigations set up by the Great Lakes Water Quality Agreement of 1972.

Section 5

NATURE AND AVAILABILITY OF TECHNOLOGY TO COPE WITH PROBLEMS

5.1 Nature of Technology

There is abundant published material on techniques of controlling pollution from mining activities (e.g., U.S. EPA Rept. EPA-430/9-73-011, 1973). Most of it deals either with acid-mine drainage and other problems of coal-mine areas, or with the treatment of waters laden with suspended solids from the milling of metallic ores. The former subject is of little significance in the Great Lakes region. The latter is applicable to the Minnesota iron-ore mines, to which indeed it has been intensively applied (Baillod et al., 1970, 1972). This technology, i.e. use of tailings ponds and related features, can be readily modified for use at gravel pits and stone quarries throughout the region if it is determined that the need exists.

When and if it is decided how best to dispose of unwanted brines, it will probably be found that existing techniques will do the job. Know-how is available to evaporate brines in order to recover their salts; to pump them into deeply buried strata via disposal wells (U. S. EPA Rept. EPA-430/9-73-012, p. 31-100); and to return them to oil-bearing strata to aid in repressuring and total ultimate recovery of oil. What is needed is fundamental research on alternative methods of disposal, especially on the long-range effects of deep-well injection.

Section 6

NEED FOR NEW PROGRAMS

6.1 Gaps in Knowledge

By far the largest extractive industries in the Great Lakes region are those that produce sand, gravel, and crushed stone. Large quantities of water are used in these operations, yet little information seems to exist on the extent of this use or on the degree to which contaminated waters are released. A systematic study, perhaps undertaken with the aid of the various states, would yield much-needed data on this subject. Such a study should be undertaken in the light of the expected expansion of these industries in the coming decades, as indicated on Table 3, page 15.

As previously indicated, there are appreciable gaps in our knowledge of the best way in which to dispose of waste brines. One aspect of the problem that should receive attention is a systematic inventory of those geologic formations beneath Michigan, Ohio, and New York that might act as large-scale receptors of brines. The rate at which these formations can accept introduced brines, and their ultimate storage capacity compared with the volume of brine expected to be produced over the next 40 to 50 years, are other topics on which information is needed.

Section 7

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Category A5 - Pesticides and Herbicides

IMPACT OF AGRICULTURAL USE OF PESTICIDES ON THE WATER QUALITY OF THE GREAT LAKES

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To be used as portion of the U. S. Task A Report on GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by the International Joint Commission - prepared in partial fulfillment of U. S. Environmental Protection Agency Contract No. 68-01-1598

July 1974

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Section 1

SUMMARY

1.1 Introduction

The agricultural land in the United States portion of the Great Lakes Region is approximately 12,998,000 ha (32,118,000 acres), representing over 38% of the total land area of the region. More than 64% of the agricultural land is cultivated and devoted to: row crops (43%), hay and pasture (33%), small grains (17%), and specialty crops (7%). Farming activities are not well-distributed in the region. Crop production is concentrated in the agricultural watersheds of Lakes Michigan and Erie where more than 75% of the total cultivated cropland is located.

Pesticides are widely used in the region. Overall, more than half of the harvested area for crop production has been treated with pesticides. Herbicides and insecticides account for most of the pesticides applied, with herbicides being the predominant type of pesticide applied. Field crops receive the major treatment of pesticides. Among field crops, row crops--particularly corn--receive substantial treatment with pesticides, while hay and pasture receive the least. Orchards are treated extensively with insecticides. Some vegetable and processing crops, such as sweet corn and potatoes, also receive considerable insecticidal treatment.

All agriculturally-applied pesticides are potential pollutants to aquatic environments. The amount and nature of pesticides reaching the Great Lakes from agricultural watersheds are primarily functions of the persistence of the compounds used, intensity and length of time pesticides have been applied, and the applicable transport mechanisms. Quantification and identification of pesticide pollution originating from agricultural activities are difficult to assess because of the complexity of the watersheds. Limited information indicates that low-level concentrations of pesticides/metabolites, particularly the persistent types, may gain entry to tributary waters and the Great Lakes essentially through agricultural runoff.

1.2 Summary of Findings and Conclusions

Fragmentary inventories of the agricultural use of pesticides indicate extensive use of pesticides in the Great Lakes Region. Herbicides are the predominant pesticides used, and their use has been increasing in recent years. The use of organophosphorous and carbamate insecticides has increased because of the need to replace the majority of the persistent organochlorine insecticides.

Accumulations of pesticides--particularly the organochlorine insecticides--have occurred in agricultural soils as a result of repeated applications. Residues of pesticides or their metabolites may find their way to aquatic environments through the atmosphere, leaching to groundwater, and runoff. Agricultural runoff has been implicated as the primary avenue of pesticide transport to aquatic environments, and many investigations, mainly on small agricultural watersheds, have been undertaken on this aspect. Data from these investigations, in addition to limited data obtained from large watersheds, indicate that only low-level residues are transported to aquatic environments via runoff. Although the amounts of residues reaching the Great Lakes are in the ng/l (ppt, i.e., 1 part in 10^{12} parts) to µg/l (ppb, 1 part in 10^{9} parts) range, contamination can be magnified by the process of bioconcentration of residues in tissues of many aquatic organisms and by accumulation in bottom sediments.

Natural dissipation processes are operating to reduce or remove pesticide residues remaining in soils and aquatic environments. Microbial metabolism appears to be the major pathway of degradation for many pesticides in soils and sediments. Intensive studies have been made on this subject, but processes are still not clearly understood and degradation products of many pesticides are not fully identified. Additionally, the fate and toxicity of degradation products need continual assessment. Most herbicides are microbially degraded in soils and are much less persistent than the organochlorine insecticides. However, many of the commonly used compounds, such as atrazine, dichlobenil, diuron, monuron, picloram, trifluralin, simazine, paraquat and diquat, may persist for a year or longer in soils. Indications have been found that the persistence of many pesticides in aquatic environments differs from that in terrestrial soils, but

further investigations are needed to make generalizations.

The pesticide pollution of the Great Lakes, particularly Lake Michigan, is already well-documented. Pesticide input to the Great Lakes will continue as long as pesticides are used in the watersheds. Projections are that the cropland area in the U.S. portion of the Great Lakes Region will remain essentially unchanged in the next 50 years, but the human population will expand rapidly, thereby necessitating increased crop production per unit area in order to meet the food requirements of the population. To sustain high crop production, pesticides are essential. Because of this consideration, in addition to the absence of economically feasible alternative methods of pest control, pesticide usage in agriculture is predicted to increase. Use of less persistent pesticides will continue to rise due to the restrictions placed on the persistent organochlorine insecticides. Even with the use of less persistent pesticides, residue build-up may still result due to increasing use and repeated applications. Accumulation and impact of the degradation products of readily degraded pesticides in the environment are relatively unknown.

The Great Lakes States appear to have adequate existing laws and regulations concerning the sale, registration, distribution, use, and application of pesticides. The primary limitations for the success of pesticide regulations are inadequate funds and lack of needed personnel to do the monitoring and enforce the programs.

The existing voluntary basis for control of erosion and sediment build-up is extremely slow and, if allowed to continue at its current rate, abatement of pesticide pollution of the Great Lakes and related waterbodies will be limited in the next 50 years. Prevention of pesticide pollution of the Great Lakes can be greatly accelerated by the enactment and enforcement of mandatory erosion- and sediment-control programs by all levels of state governments concerned.

1.3 Recommendations

Research is needed urgently to adequately quantify and identify the contribution of agricultural activities to the pesticide pollution of the Great Lakes. Presently, only fragmentary information is available, and

the impact of agricultural usage of pesticides on the water quality of the Great Lakes can be only roughly estimated. Action should be taken on the following recommendations to provide meaningful information to assess the impact on water quality of pesticide usage for agricultural purposes.

a. Accounting of pesticide inputs to agricultural lands involving a systematic documentation of the kinds and amounts of pesticides used, category of crops treated, and area treated.

b. Evaluation of degradation rates of pesticides, particularly the newer compounds, including the formation, fate, and toxicity of their degradation products.

c. Investigations on the mechanisms of pesticide transport from varied soil types and cropping patterns present in the watersheds. Atmospheric inputs of pesticides to the Great Lakes should be investigated.

d. Expansion and redesigning of monitoring programs to include soil, sediment, and atmospheric samplings. Critical evaluation of sampling techniques, sample preservation, and analytical techniques should be made as often as possible.

e. Enactment of mandatory erosion- and sediment-control regulations by all levels of state governments. If regulations are already enacted, immediate enforcement is needed.

f. Better implementation of pesticide management techniques compatible with efficient soil and crop management systems must be practiced.

Section 2

SUPPORTING MATERIAL

2.1 Introduction

Agricultural activities in the Great Lakes Basin were initiated at the beginning of the 19th century, but the use of pesticides in crop production was virtually nonexistent prior to the mid-1940's. For the past three decades pesticide usage has vastly increased, making it possible to obtain the high yields of many crops never realized before. Pesticide use patterns indicate that the agricultural industry will not only continue to use pesticides, but it is predicted that they will be used at higher than present levels in order to sustain the higher crop production per unit area needed to meet the food demands of the rapidly expanding population around the Great Lakes. Since the early 1960's public concern has been expressed that pesticide residues--particularly the organochlorine insecticides--are widely distributed in the biosphere and have accumulated in several segments of the environment including the Great Lakes. Farming operations, which provide the major use for pesticides, are frequently cited for the pesticide contamination of the aquatic environment. These suspicions, however, have not been fully documented, and an attempt is made in this review to evaluate the present and future impact on the water quality of the Great Lakes from pesticides used on agricultural land.

2.2 Scope of Study

Data were obtained from published papers, reports, symposia proceedings, framework studies, abstracts, texts, and personal communications. References from the period of 1960 to 1974 were considered, but most of the literature surveyed was published and/or reported in the late 1960's and early 1970's. Although much of the information included was gathered outside the Great Lakes Basin, a fair amount of information is available for the basin and is given emphasis in the report.

2.3 Study Procedure

The literature evaluation on the sources, forms, and amounts of pesticide pollutants arising from agricultural activities was conducted under the direct supervision of the Director of the University of Wisconsin Water Resources Center (G. Chesters). A graduate student (G. V. Simsiman) assisted in compiling and summarizing needed data and wrote the preliminary report. Final review and revision of the draft submitted to the Great Lakes Basin Commission were done by the Director.

Information was gathered by writing many scientists, institutions, and government agencies, requesting copies of reprints, preprints, reports, and other kinds of information. The search for ongoing research on pesticides was undertaken utilizing the services of the University of Oklahoma-GIPSY computer program which is supported by the Office of Water Resources Research of the U.S. Department of the Interior. The major sources of literature/information used were:

1. Journal of Environmental Quality
2. Pesticide Monitoring Journal
3. Soil Science Society of America Proceedings
4. Journal of Economic Entomology
5. Journal of Agricultural and Food Chemistry
6. Weed Science
7. Weed Research
8. Environmental Science and Technology
9. Water Resources Bulletin
10. BioScience
11. Science
12. Water Pollution Control Federation Journal
13. Agronomy Journal
14. Residue Reviews
15. Advances in Agronomy
16. Proceedings of CIC-CCIW Symposium on Water Quality Parameters--Selection, Detection, and Measurement, Burlington, Ontario, November 1973

17. Proceedings of Seminar on Environmental Toxicology of Pesticides, Academic Press, N.Y.

18. Proceedings of Second International IUPAC Congress of Pesticide Chemistry, Gordon and Breach Publishers, N.Y.

19. Proceedings of Symposium on the Fate of Organic Pesticides in the Aquatic Environment, American Chemical Society, Washington, D.C.

20. Proceedings of Symposium on Organic Pesticides in the Environment, American Chemical Society, Washington, D.C.

21. Proceedings of Symposium on Agriculture and the Quality of Our Environment, AAAS, Washington, D.C.

22. Proceedings of Conference on Chemical Fallout, Current Research on Persistent Pesticides, Charles C. Thomas, Springfield, Illinois

23. Proceedings Cornell University Conference on Agricultural Waste Management, Ithaca, N.Y., February 1971

24. Monograph on the Degradation of Herbicides

25. Proceedings on Pesticides, Pest Control and Safety on Forest and Range Lands, A Continuing Education Book, Corvallis, Oregon

26. Chemical Rubber Company Monoscience Series

27. Report of the Secretary's Commission on Pesticides and Their Relationship to Environmental Health, U.S. Department of Health, Education and Welfare

28. Proceedings of Conferences on Great Lakes Research

29. Great Lakes Basin Framework Study

30. International Joint Commission Annual Reports

31. Selected Water Resources Abstracts

32. Water Resources Research Catalogs

33. USDA-CSRS Current Research Information Service

34. USDI, OWRR, Water Resources Scientific Information Center

35. USDA Agricultural Stabilization Service

36. USDA Economic Research Service

37. U.S.EPA Ecological Research Series

38. U.S.EPA Water Pollution Control Research Series

39. U.S.EPA Environmental Protection Technology Series

40. U.S.EPA Office of Pesticide Programs

41. Water Resources Center, Purdue University

42. Water Resources Center, University of Wisconsin-Madison

43. Water Resources Information Program, University of Wisconsin-Madison

44. Department of Entomology, University of Wisconsin-Madison
45. Michigan Department of Natural Resources
46. Wisconsin Department of Agriculture
47. Wisconsin Department of Natural Resources
48. Department of Soil Science, University of Wisconsin-Madison
49. Water Chemistry Degree Program, University of Wisconsin-Madison

2.3.1 Assumptions

Pesticide contamination of the Great Lakes has been revealed by state and federal monitoring programs. However, there are very few well-documented investigations on the sources, forms, and amounts of pesticide pollutants from various segments of the watersheds, particularly those from agricultural lands. The following assumptions are made:

a. The use of pesticides is inevitable and will continue to increase, particularly in the case of herbicides, because of the present lack of feasible alternatives for pest control.
b. Increased production per unit area on the essentially unchanged area devoted to crop production in the Great Lakes Region is necessary to meet the food requirements of the rapidly expanding population therein. To produce such high crop yields more and better production inputs, including pesticides, are needed.
c. Accumulated residues of pesticides, resulting from repeated applications to agricultural soils, serve as reservoirs for potentially transportable residues.
d. Soil erosion is a continuous geologic process but is accelerated by farming activities. Agricultural runoff, laden with pesticides, eventually reaches surface waters and provides low-level concentrations of residues. Abatement of soil erosion is proceeding only slowly and in a limited fashion.
e. The impact on water quality of agricultural use of pesticides is great in the Great Lakes Basin due to the extensive farming activities in the basin and the vast water surface area involved.
f. Agricultural use of pesticides in the Great Lakes Basin produces similar processes and relationships to those reported in other areas of the nation.

Section 3

REVIEW OF FINDINGS OF PREVIOUS STUDIES

3.1 General Description of Agricultural Land Use Category

3.1.1 Agricultural Activities

The Great Lakes Region consists of a total land area of 33,824,200 ha (83,579,700 acres) (66). In 1967 there were 11,579,000 ha (28,612,000 acres) of cropland; 1,419,000 ha (3,506,000 acres) of pasture land; 16,035,900 ha (39,624,700 acres) of forest land; 2,827,900 ha (6,987,700 acres) of urban and built-up areas; and 1,963,800 ha (4,852,500 acres) of other land. Cropland, including pasture, comprises the agricultural sector, and this sector represents more than 38% of the land area of the region. Forestry is practiced on 47% of the land; urban and built-up areas cover 8.4%; while 5.8% of the land is given over to other uses.

A wide variety of crops is grown in the region, including small grains, row crops, hay, vegetables, and fruits (Table 1). Estimates indicate that crop production is sufficient to supply the food and feed requirements of the population and the livestock and dairy industry of the region. The total cultivated cropland in 1967 comprised 8,361,700 ha (20,661,700 acres) or over 64% of the total agricultural land (includes cultivated cropland, pasture, and idle cropland). Over 43% of the cultivated cropland is devoted to row crops; 33% to hay and pasture; 17% to overall grains; and 7% to specialty crops (Table 2).

In order to sustain efficient crop production, good management practices are needed. Pest and disease control by pesticides has become indispensable in many areas of crop production.

Farm types and their location in certain geographic areas are related to climate, soils and markets. Farming activities appear to be concentrated in the Lake Michigan and Lake Erie regions (Table 2). About 75% of the total area devoted to producing all crop categories lies in these regions

TABLE 1

Land area and crop production in metric and English units for 1967 and predictions for 1980, 2000 and 2020 for the U.S. Great Lakes Region. (Recalculated from Appendix 13, Land Use and Management Work Group of the Great Lakes Basin Commission sponsored by the Forest Service and Soil Conservation Service of the U.S. Department of Agriculture (66).)

Crop	1967				1980				2000				2020			
	Area		Production		Area		Production		Area		Production		Area		Production	
	Hax10^3	Ax10^3	MTx10^3	Unit**x10^3	Hax10^3	Ax10^3	MTx10^3	Unit**x10^3	Hax10^3	Ax10^3	MTx10^3	Unit**x10^3	Hax10^3	Ax10^3	MTx10^3	Unit**x10^3
Wheat	710	1,756	1,773	65,142 Bu	797	1,970	2,624	96,400 Bu	754	1,862	3,044	111,800 Bu	796	1,967	3,637	133,600 Bu
Oats	686	1,696	1,454	100,135 Bu	756	1,868	2,241	154,375 Bu	649	1,604	2,083	143,500 Bu	423	1,046	1,536	105,813 Bu
Rye	24	60	44	1,739 Bu	24	59	48	1,902 Bu	28	69	64	2,516 Bu	35	87	86	3,386 Bu
Barley	18	45	45	2,089 Bu	39	96	129	5,917 Bu	35	86	132	6,042 Bu	29	71	122	5,625 Bu
Misc. Small Grains	17	43														
Corn for Grain	1,768	4,369	8,887	349,759 Bu	1,469	3,630	9,740	383,357 Bu	1,606	3,969	12,874	506,679 Bu	1,965	4,856	17,152	675,071 Bu
Corn for Silage	494	1,221	13,577	14,962 Ton	363	897	14,858	16,374 Ton	409	1,010	19,877	21,904 Ton	518	1,279	26,236	28,912 Ton
Soybeans	1,054	2,604	1,781	65,426 Bu	1,387	3,428	2,925	107,440 Bu	1,382	3,416	3,676	135,048 Bu	1,524	3,766	4,742	174,176 Bu
Dry Edible Beans	306	756	288	6,352 CWT	295	730	516	11,376 CWT	331	819	682	15,029 CWT	402	993	915	20,171 CWT
Sugar Beets	52	128	1,451	1,599 Ton	90	222	2,920	3,218 Ton	98	241	4,791	5,280 Ton	105	259	7,421	8,178 Ton
Potatoes	61	152	816	17,987 CWT	29	71	961	21,180 CWT	28	70	1,315	28,988 CWT	36	90	1,833	40,404 CWT
Fruits	243	600	1,002	1,104 Ton	161	398	1,323	1,458 Ton	165	407	1,903	2,097 Ton	185	456	2,719	2,996 Ton
Commercial Vegetables	211	520	2,091	46,093 CWT	213	526	3,284	72,380 CWT	226	558	4,505	99,295 CWT	267	659	6,224	137,171 CWT
Commercial Sod	21	53			21	53			21	53			21	53		
Alfalfa Hay	1,497	3,699	8,159	8,991 Ton	1,131	2,794	8,859	9,763 Ton	991	2,450	9,150	10,084 Ton	979	2,420	9,966	10,982 Ton
Clover-Tim-Other Hay	778	1,921	2,786	3,070 Ton	538	1,330	3,072	3,385 Ton	433	1,070	2,812	3,099 Ton	363	897	2,739	3,018 Ton
Cropland Pasture	422	1,042			303	749	1,312 *	1,446 *	270	667	1,500 *	1,653 *	289	714	2,077 *	2,289 *
Idle Cropland	3,216	7,947	0	0	3,618	8,939	0	0	3,316	8,194	0	0	2,396	5,920	0	0
TOTAL CROPLAND	11,578	28,612			11,234	27,758			10,742	26,542			10,333	25,533		
Improved Pasture	378	934			368	909	1,744 *	1,922 *	353	872	2,007 *	2,212 *	340	839	2,180 *	2,402 *
Improvable Pasture	908	2,243			888	2,194	3,844 *	4,236 *	859	2,123	2,476 *	2,729 *	833	2,058	2,734 *	3,013 *
Non-Improvable Pasture	133	329			131	325	167 *	184 *	128	317	188 *	207 *	128	316	210 *	231 *
TOTAL PASTURE	1,419	3,506			1,387	3,428			1,340	3,312			1,301	3,212		
TOTAL AGRICULTURAL LAND	12,997	32,118			12,621	31,186			12,082	29,854			11,634	28,745		

*Alfalfa hay equivalents (tons)
**Bushel weights for wheat, oats, rye, barley, miscellaneous small grains, corn for grain, and soybeans are 60, 32, 56, 48, 56, 56, and 60 lb, respectively.

Ha = Hectare
A = Acre
MT = Metric Ton

TABLE 2

Total area cultivated and area cultivated by crop categories for 1967 and projections for 1980, 2000, and 2020 in the U.S. Great Lakes Region. (Modified and recalculated from Appendix 13, Land Use and Management Work Group of the Great Lakes Basin Commission, sponsored by the Forest Service and Soil Conservation Service of the U.S. Department of Agriculture (66).)

Region	1967		1980		2000		2020	
	$Ha x 10^3$	$A x 10^3$	$Ha x 10^3$	$A x 10^3$	$Ha x 10^3$	$A x 10^3$	$Ha x 10^3$	$A x 10^3$
				TOTAL CROPLAND*				
Lake Superior	156.9	392.6	117.1	289.4	83.0	205.2	83.2	205.5
Lake Michigan	3,681.7	9,097.5	3,422.9	8,458.1	3,342.5	8,259.3	3,614.8	8,932.2
Lake Huron	825.2	2,039.1	810.1	2,001.8	875.1	2,162.3	1,007.8	2,490.2
Lake Erie	2,726.3	6,736.6	2,574.1	6,360.7	2,548.4	6,297.1	2,612.9	6,456.4
Lake Ontario	969.5	2,395.7	691.6	1,709.0	592.2	1,463.4	618.7	1,528.7
TOTAL	8,361.7	20,661.7	7,615.9	18,819.0	7,441.2	18,347.3	7,937.2	19,612.9
				SPECIALTY CROPS**				
Lake Superior	3.8	9.4	1.5	3.6	1.2	2.9	1.3	3.2
Lake Michigan	283.5	700.6	226.4	559.5	240.9	595.2	279.8	691.4
Lake Huron	46.5	115.0	70.2	173.5	79.0	195.2	90.6	223.8
Lake Erie	136.5	337.3	118.0	291.5	96.3	238.0	137.8	340.4
Lake Ontario	94.8	234.3	142.0	351.0	77.5	191.5	84.2	208.2
TOTAL	566.8	1,400.7	492.7	1,217.4	516.2	1,275.5	592.8	1,464.8
				ROW CROPS**				
Lake Superior	1.4	3.4	0.77	1.9	1.2	3.1	2.3	5.6
Lake Michigan	1,506.0	3,721.3	1,447.0	3,575.5	1,522.4	3,761.9	1,848.2	4,567.0
Lake Huron	398.1	983.7	361.6	893.4	413.7	1,022.2	518.5	1,281.2
Lake Erie	1,532.0	3,785.7	1,571.9	3,884.2	1,670.4	4,127.7	1,887.2	4,663.2
Lake Ontario	184.9	457.0	132.8	328.2	121.1	299.2	152.7	377.3
TOTAL	3,622.0	8,949.9	3,514.4	8,684.0	3,729.0	9,214.4	4,408.9	10,894.4
				SMALL GRAINS**				
Lake Superior	18.6	46.0	24.4	60.4	12.3	30.3	12.2	30.2
Lake Michigan	646.2	1,596.9	753.2	1,861.3	712.9	1,761.7	666.5	1,646.9
Lake Huron	153.6	379.5	202.8	501.0	197.6	488.3	190.1	469.8
Lake Erie	476.9	1,178.5	468.2	1,157.0	210.2	519.5	288.7	713.4
Lake Ontario	161.2	398.4	166.9	412.4	129.5	320.0	125.3	309.7
TOTAL	1,456.6	3,599.3	1,615.9	3,992.8	1,465.4	3,621.1	1,283.0	3,170.4
				HAY AND PASTURE**				
Lake Superior	135.1	333.8	90.4	223.5	68.4	168.9	67.4	166.5
Lake Michigan	1,245.7	3,078.2	996.3	2,461.8	866.2	2,140.5	820.3	2,027.0
Lake Huron	227.0	560.9	175.6	433.9	184.8	456.7	208.6	515.4
Lake Erie	580.8	1,435.1	416.0	1,028.0	331.2	818.5	300.3	742.0
Lake Ontario	528.4	1,305.8	315.3	779.1	264.1	652.7	256.4	633.5
TOTAL	2,716.2	6,711.8	1,993.0	4,924.8	1,714.4	4,236.3	1,652.5	4,083.3

*Excludes idle cropland

**Specialty crops: fruits, vegetables, potatoes, sugar beets; Row crops: corn, corn silage, soybeans, dry beans; Small grains: wheat, oats, barley, rye, misc. small grains; Hay and pasture: alfalfa, clover-timothy, cropland pasture, sod.

of Lakes Michigan and Erie. Based on the total cultivated cropland in 1967, crop production in terms of area in the five lake regions is in the order: Lake Michigan (43%) >Lake Erie (33%) >Lake Ontario (12%) >Lake Huron (10%) >Lake Superior (2%). Higher intensity of crop production in the Lake Michigan and Lake Erie regions implies that more pesticides have been used in these regions than in other lake regions.

3.1.2 Pesticide Use Patterns

In assessing the extent by which pesticides contaminate the environment it is important to determine how much has been used. Estimates of quantities of pesticides used by farms in 1966 (183) indicated that 160,000,000 kg (353,000,000 lb) of active ingredients were used for crops, livestock, and other purposes, representing 51% of the total quantity used in the United States which was 310,000,000 kg (681,000,000 lb). Over 149,000,000 kg (328,000,000 lb) were used for crops alone, which includes all crops, pasture and rangeland.

According to the Census of Agriculture, 1969 (184) about 16,000,000 ha (40,000,000 acres) of agricultural land in the eight Great Lakes States were treated with pesticides, comprising 28% of the total area treated in the United States. Pesticide usage in the Great Lakes Region and neighboring areas indicated that herbicides are the leading types of pesticides (>50%) used in treated areas followed by insecticides (183, 184, 205). Use of other pesticides, such as nematocides, fungicides and defoliants, are not as widely used as herbicides and insecticides. Herbicide use has been rapidly increasing, while use of other pesticides has not changed substantially.

An inventory of the kinds and quantities of pesticides used in the Great Lakes Region was first attempted in 1966 (43). According to this survey, 12,421,960 kg (27,378,000 lb) of pesticides (active ingredients, including petroleum and sulfur) were applied, with crops (all crops, pasture and rangeland) receiving the largest share (86%), followed by livestock (12%), and other purposes (2.0%). The 9,431,942 kg (20,788,000 lb)

of pesticides used in the region, exclusive of petroleum and sulfur, represent 6.3% of the 149,000,000 kg (328,000,000 lb) used on crops in the United States. A vast array of chemicals used on crops is listed in Tables 3, 4, 5 and 6. It is significant to note from the list that during this period herbicides constituted 51% of the total used, followed by insecticides (32%), fungicides (15%), and miscellaneous pesticides (1.7%). Of the total insecticides applied, the organochlorine insecticides (57%) were predominant over the organophosphorous compounds (20%) and carbamate insecticides (11%). The most widely used organochlorine compounds were aldrin, DDT, heptachlor and toxaphene. Corn was the major recipient of herbicides and insecticides. Apple and vegetable crops received substantial amounts of insecticides and fungicides.

Pesticide surveys were conducted after 1966 in five Great Lakes States on field crops (205) and in Wisconsin on fruits, vegetables, and processing crops (108). These surveys were not specific for the Great Lakes Region; however, they reflect changes in usage of pesticides throughout the area, particularly in Michigan and Wisconsin.

The study concerning the general farm use of pesticides was recommended by a task force created by the Governors of Illinois, Indiana, Michigan, Minnesota and Wisconsin after a conference in April, 1969 to review the pesticide situation at that time as it related to possible pesticide pollution. Primary emphasis was placed on the major field crops which account for the largest share of pesticide usage. The survey was conducted for 3 years (1969 to 1971) in all states of the group except Indiana which did not conduct the survey in 1971. In 1971, approximately 18,900,000 ha (46,800,000 acres) of field crops were treated with pesticides in the four lake states, an increase of 18% over 1969. Herbicides accounted for 79% of the treated area, with insecticides applied to nearly 20%. Less than 1% of the area received fungicides. Herbicide usage increased by 28% from 1969 to 1971, while use of insecticides decreased by 10%.

About 13,400,000 ha (33,200,000 acres) or 62% of the harvested area of 21,700,000 ha (53,600,000 acres) received herbicide and/or insecticide treatment at least once in 1971, which is 8% more than in 1969 (Table 7).

TABLE 3

Quantities used and area treated with selected fungicides, all crops, U.S. Great Lakes Basin, 1966.* (Adapted from Cywin and Ward (43).)

Fungicide**	Active Ingredients		Area Treated+	
	kgx10^3	lbx10^3	Hax10^3	Ax10^3
Inorganic				
Copper sulfates	11.4	25.2	4.4	11.0
Other coppers	57.3	126	14.4	35.6
Mercury compounds	2.6	5.8	11.7	28.8
Other inorganics	40.0	88.2	5.8	14.4
TOTAL INORGANICS (NOT INCLUDING SULFUR)	111	245		
Organic				
Dithiocarbamates				
Maneb	278	613	35.9	88.6
Zineb	382	842	87.2	215
Ferbam	70	154	18.5	45.8
Others	58	128	9.1	22.4
TOTAL DITHIOCARBAMATES	788	1,737		
Phthalimides				
Captan	395	871	59.9	148
Others	21	47	3.9	9.6
TOTAL PHTHALIMIDES	416	918		
Karathane, Dodine, Quinones	78.5	173	33.2	82.0
Phenols	7.0	15.4	2.4	6.0
Other organics	13.8	30.4	7.0	17.2
TOTAL ORGANICS	1,304	2,874		
TOTAL (NOT INCLUDING SULFUR)	1,415	3,119		
Sulfur	501	1,105	26.3	65.0
TOTAL FUNGICIDES	1,916	4,224		

*Estimates based on use shown by the ERS Pesticide and General Farm Survey, 1966.

**May include use for purposes other than as fungicides.

+Not additive since one or more ingredients or different commercial preparations of a single ingredient may be applied on same land areas.

TABLE 4

Quantities used and area treated with selected herbicides, all crops, U.S. Great Lakes Basin, 1966.* (Adapted from Cywin and Ward (43).)

Herbicide**	Active Ingredients		Area Treated†	
	kgx10^3	lbx10^3	Hax10^3	Ax10^3
Inorganic	110	243	19.6	48.4
Organic				
Arsenicals	1.5	3.4	2.4	6.0
Phenoxy				
2,4-D	1,202	2,650	1,681	4,153
2,4,5-T	21.1	46.6	45.0	111
MCPA	45.2	100	89.8	222
Other	2.0	4.4	9.6	23.8
TOTAL PHENOXY	1,270	2,801		
Phenyl urea				
Diuron	10.2	22.6	8.4	20.8
Linuron	116	256	78.0	193
Other	5.2	11.4	5.0	12.4
TOTAL PHENYL UREA	131	290		
Amides				
Propachlor	201	443	125	309
Propanil	---	---	---	---
NPA	67.6	149	48.0	119
TOTAL AMIDES	269	592		
Carbamates				
CIPC and IPC	35.0	77.2	32.5	80.2
CDAA	432	953	294	725
Other	126	278	37.8	93.4
TOTAL CARBAMATES	593	1,308		
Dinitro group	114	252	35.4	87.6
Triazines				
Atrazine	1,542	3,398	864	2,136
Propazine	---	---	---	---
Other	16.3	36.0	8.1	20.0
TOTAL TRIAZINES	1,558	3,434		
Benzoic				
2,3,6-TBA	249	549	118	292
Amiben	323	713	309	763
Dicamba	3.8	8.4	10.9	27.0
TOTAL BENZOIC	576	1,270		
Other organics				
Trifluralin	108	237	105	262
Others	103	226	63.0	156
TOTAL OTHER ORGANICS	211	463		
TOTAL ORGANICS (NOT INCLUDING PETROLEUM)	4,725	10,413		
TOTAL HERBICIDES (NOT INCLUDING PETROLEUM)	4,835	10,656		
Petroleum	573	1,264	24.8	61.4
TOTAL HERBICIDES (INCLUDING PETROLEUM)	5,408	11,920		

*Estimates based on use shown by the ERS Pesticide and General Farm Survey, 1966.

**May include use for purpose other than as herbicides.

†Not additive since one or more ingredients or different commercial preparations of a single ingredient may be applied on same land areas.

TABLE 5

Quantities used and areas treated with selected insecticides, all crops, U.S. Great Lakes Basin, 1966.* (Adapted from Cywin and Ward (43).)

Insecticides**	Active Ingredients		Area Treated+	
	kgx10^3	lbx10^3	Hax10^3	Ax10^3
Inorganics	358	790	22.1	54.6
Botanicals and biologicals	0.3	0.6	2.7	6.8
Synthetic organics				
Organochlorines				
Lindane	11.0	24.2	33.3	82.4
Strobane	---	---	---	---
TDE (DDD)	44.9	99.0	11.8	29.2
DDT	138	305	72.7	180
Methoxychlor	39.4	86.8	36.4	90.0
Endrin	0.5	1.0	1.0	2.4
Heptachlor	133	292	153	378
Dieldrin	16.2	35.8	20.6	51.0
Aldrin	1,245	2,744	1,033	2,552
Chlordane	26.0	57.2	18.9	46.6
Endosulfan	14.0	30.8	11.2	27.8
Toxaphene	47.0	104	26.1	64.4
Others	0.4	0.8	1.5	3.8
TOTAL ORGANOCHLORINE	1,715	3,780		
Organophosphorus				
Disulfoton	67.4	149	47.2	117
Bidrin	---	---	---	---
Methyl parathion	6.8	15.0	0.8	2.0
Parathion	110	242	128	317
Malathion	50.4	111	36.3	89.8
Diazinon	230	507	216	533
Trichlorfon	0.2	0.4	0.1	0.2
Azinphosmethyl	72.5	160	50.0	124
Ethion	32.7	72.0	14.1	34.8
Others	30.2	66.6	27.7	68.4
TOTAL ORGANOPHOSPHORUS	600	1,323		
Carbamates				
Carbaryl	306	674	82.3	203
Others	16.1	35.4	24.8	61.4
TOTAL CARBAMATES	322	709		
Other synthetic organics	1.1	2.4		
TOTAL SYNTHETIC ORGANICS	2,639	5,816		
TOTAL INSECTICIDES (NOT INCLUDING PETROLEUM)	2,997	6,606		
Petroleum	177	390	7.4	18.4
TOTAL INSECTICIDES	3,175	6,997		

*Estimates based on use shown by the ERS Pesticide and General Farm Survey, 1966.

**May include use for purposes other than as insecticides.

+Not additive since one or more ingredients or different commercial preparations of a single ingredient may be applied on same land area.

TABLE 6

Quantities used and areas treated with selected miscellaneous pesticides, all crops, U.S. Great Lakes Basin, 1966.* (Adapted from Cywin and Ward (43).)

Miscellaneous Pesticides**	Active Ingredients		Area Treated+	
	kgx10^3	lbx10^3	Hax10^3	Ax10^3
Miticides				
Dicofol	21.3	47.0	11.9	29.4
Chlorobenzilate	0.9	2.0	0.7	1.6
Aramite	---	---	---	---
Tetradifon	15.9	35.0	12.1	30.0
Others	6.2	13.6	7.7	19.0
TOTAL MITICIDES	44.3	97.6		
Fumigants				
Nemagon	10.0	22.0	0.1	0.2
D-D mixture	11.0	24.2	0.1	0.2
Sulfur dioxide	43.4	95.6	3.9	9.6
Others	49.9	110	27.0	66.8
TOTAL FUMIGANTS	114	252		
Defoliants and dessicants				
Arsenic acid	---	---	---	---
Magnesium chlorate	---	---	---	---
DEF and Folex	7.5	16.6	4.9	12.0
Others	---	---	---	---
TOTAL DEFOLIANTS AND DESSICANTS	7.5	16.6		
Rodenticides	0.8	1.8	3.0	7.4
Plant growth regulators				
Maleic hydrazide	14.5	32.0	1.8	4.4
Others	0.1	0.2	0.5	1.2
TOTAL PLANT GROWTH REGULATORS	14.6	32.2		
Repellents	2.8	6.2	1.1	2.6
TOTAL MISCELLANEOUS PESTICIDES	184	406		

*Estimates based on use shown by the ERS Pesticide and General Farm Survey, 1966.

**May include use for purposes other than those indicated.

+Not additive since one or more ingredients or different commerical preparations of a single ingredient may be applied on same land areas.

TABLE 7

Estimated areas of crops harvested and crops treated with herbicides for five Great Lakes States during 1969, 1970 and 1971.* (Recalculated from General Farm Use of Pesticides 1971 Wisconsin and Illinois, Michigan and Minnesota, Wisconsin Statistical Reporting Service, sponsored by the Wisconsin Department of Agriculture and U.S. Department of Agriculture (205).)

Harvested or Treated	Year	Illinois		%	Indiana		%	Michigan		%	Minnesota		%	Wisconsin		%	Total*		%
		Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated
								CORN											
Harvested	1969	4,039	9,980		1,983	4,901		673	1,662		1,999	4,939		1,079	2,666		7,789	19,247	
	1970	4,148	10,250		2,102	5,195		720	1,778		2,139	5,285		1,111	2,746		8,118	20,059	
	1971	4,229	10,450		--	--		902	2,230		2,644	6,533		1,233	3,048		9,009	22,261	
Treated	1969	3,413	8,433	84	1,727	4,267	87	585	1,445	87	1,524	3,765	76	832	2,055	77	6,353	15,698	82
	1970	3,510	8,673	85	1,807	4,464	86	642	1,587	89	1,706	4,216	80	967	2,390	87	6,826	16,866	84
	1971	3,722	9,196	88	--	--		824	2,036	91	2,231	5,513	84	1,064	2,630	86	7,841	19,375	87
								SMALL GRAINS+											
Harvested	1969	816	2,016		494	1,221		465	1,149		2,017	4,985		722	1,784		4,020	9,934	
	1970	666	1,645		442	1,093		444	1,097		1,969	4,866		719	1,777		3,798	9,385	
	1971	659	1,628		--	--		420	1,039		2,233	5,518		661	1,634		3,974	9,819	
Treated	1969	8	21	1	19	48	4	147	364	31	1,212	2,994	60	150	370	21	1,517	3,749	38
	1970	22	55	3	20	50	5	156	385	35	1,105	2,732	56	170	420	24	1,454	3,592	38
	1971	59	147	9	--	--		166	410	40	1,563	3,863	70	162	400	25	1,951	4,820	49
								SOYBEANS											
Harvested	1969	2,724	6,730		1,340	3,311		208	514		1,242	3,068		70	174		4,244	10,486	
	1970	2,752	6,800		1,340	3,311		212	524		1,266	3,129		62	153		4,292	10,606	
	1971	2,894	7,150		--	--		219	540		1,154	2,851		52	128		4,318	10,669	
Treated	1969	1,907	4,711	70	923	2,280	69	118	292	57	690	1,706	56	35	87	50	2,750	6,796	65
	1970	1,970	4,867	72	951	2,350	71	134	330	63	783	1,936	62	31	77	50	2,918	7,210	68
	1971	2,289	5,656	79	--	--		154	380	70	793	1,960	64	24	59	46	3,260	8,055	76
								HAY											
Harvested	1969	503	1,243		387	956		601	1,485		1,350	3,336		1,628	4,022		4,082	10,086	
	1970	510	1,260		377	932		577	1,425		1,308	3,231		1,625	4,016		4,019	9,932	
	1971	494	1,220		--	--		587	1,450		1,315	3,250		1,653	4,086		4,049	10,006	
Treated	1969	30	75	6	47	117	12	101	250	17	19	48	1.4	8	20	0.5	159	393	4
	1970	23	57	5	25	61	7	98	242	17	8	20	0.6	9	22	0.6	138	341	3
	1971	22	55	5	--	--		73	181	12	5	13	0.4	11	28	0.7	112	277	3
								MISCELLANEOUS CROPS++											
Harvested	1969	--	--		--	--		272	671		--	--		3	7		274	678	
	1970	--	--		--	--		239	590		165	407		4	9		407	1,006	
	1971	--	--		--	--		239	590		97	240		4	10		340	840	
Treated	1969	--	--		--	--		200	493	74	--	--		1	3	43	201	496	74
	1970	--	--		--	--		196	484	82	113	280	69	1	3	33	310	767	76
	1971	--	--		--	--		205	507	86	58	143	60	3	7	70	266	657	78
								TOTAL											
Harvested	1969	8,081	19,969		4,204	10,389		2,218	5,481		6,608	16,328		3,502	8,653		20,409	50,431	
	1970	8,076	19,955		4,262	10,531		2,191	5,414		6,847	16,918		3,521	8,701		20,635	50,988	
	1971	8,275	20,448		--	--		2,367	5,849		7,443	18,392		3,604	8,906		21,690	53,595	
Treated	1969	5,358	13,240	66	2,716	6,712	65	1,151	2,844	52	3,445	8,513	52	1,026	2,535	29	10,980	27,132	54
	1970	5,525	13,652	68	2,803	6,925	66	1,225	3,028	56	3,717	9,184	54	1,178	2,912	33	11,645	28,776	56
	1971	6,092	15,054	74	--	--		1,422	3,514	60	4,651	11,492	62	1,264	3,124	35	13,429	33,184	62

*Area treated with both herbicide and insecticide or more than once with either are counted only once.

**Does not include Indiana because of lack of data in 1971

+Includes oats, wheat, barley and rye

++Dry beans in Michigan, flax in Minnesota and tobacco in Wisconsin

Corn received 58% of the treatment, soybeans 24%, small grains 14%, and hay less than 1%. Large portions of the harvested area of corn (87%), soybeans (75%), and small grains (50%) were treated. Less than 3% of the hay area received any pesticide.

Atrazine, alone or in combination, and propachlor were the herbicides most commonly used on corn both for pre-emergence and post-emergence application (Table 8). The amine and ester formulations of 2,4-D were extensively used for post-emergence treatment of corn. Amiben was used on almost half of the soybeans treated. MCPA and 2,4-D amine were the leading herbicides applied to small grains.

Most of the insecticides were used on corn largely as pre-emergence treatments (Table 9). Only small portions of the soybean, small grain, and hay area were treated for insects. Aldrin, carbofuran, phorate, diazinon and buxten were the major insecticides used on corn. Major insecticides used for other field crops were of the organophosphorous and carbamate types.

A sharp decline in the usage of organochlorine insecticides was reported in the four Great Lakes States for 1971. Less than 1,400,000 ha (3,500,000 acres) of field crops (6% of the treated area) were treated, a decrease of 20% from 1970. Aldrin, used primarily on corn, topped all other organochlorine insecticides, with heptachlor and chlordane being used to lesser extents. The usage of DDT on field crops in 1971 continued to be minimal.

The use of insecticides on vegetables, fruits and crops for processing in Wisconsin was surveyed in 1969 (108). Approximately 83,925 ha (207,380 acres) or 51% of the total area of vegetables and crops for processing and 9,841 ha (24,317 acres) or 97% of the total fruit area were treated with insecticides. Of the 463,880 kg (1,022,385 lb) total insecticides applied in 1969, about 65% was used for vegetables and crops for processing. Sweet corn, potatoes, and apples received major portions of the insecticides.

In the Lake Michigan region of Wisconsin--which constitutes 31% of the insecticide-treated area of the state--approximately 177,092 kg (390,310 lb) or 38% of the total insecticides was used. A wide variety of chemicals were applied (Table 10). Lead arsenate, carbaryl, phorate, malathion, parathion, azinphosmethyl, and toxaphene, in that order, were the leading insecticides used.

TABLE 8

Rate of application and estimated area treated with major herbicides in four Great Lakes States for 1971. (Recalculated from General Farm Use of Pesticides 1971 Wisconsin and Illinois, Michigan and Minnesota, Wisconsin Statistical Reporting Service, sponsored by the Wisconsin Department of Agriculture and U.S. Department of Agriculture (205).)

Herbicide	Formulation*	Average rate of application per unit area**								Estimated area treated*							
		Illinois		Michigan		Minnesota		Wisconsin		Illinois		Michigan		Minnesota		Wisconsin	
		kg or liter	lb or qt	kg or liter	lb or qt	kg or liter	lb or qt	kg or liter	lb or qt	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3
CORN - PREEMERGENCE																	
All herbicides	WP	1.1	2.4	1.0	2.2	0.87	1.9	0.91	2.0								
	GP	3.3	7.3			4.0	8.7	2.1	4.7	3,310	8,178	507	1,252	1,339	3,308	735	1,815
	LC	1.5	1.6	1.2	1.3	1.4	1.5	1.6	1.7								
Atrazine	WP	1.0	2.2	1.1	2.3	0.82	1.8	0.96	2.1	788	1,946	353	872	349	862	523	1,293
Atrazine and propachlor	WP	1.9	4.2			1.1	2.5	1.4	3.0								
	GP	3.1	6.7			3.4	7.4			189	466			67	166	5.7	14
Atrazine and alachlor	WP	0.73	1.6	0.69	1.5	0.78	1.7	0.78	1.7								
	LC	1.8	1.9	1.4	1.5			1.6	1.7	126	311	92	227	30	73	105	260
Atrazine and sutan	GP	0.78	1.7	0.69	1.5	0.73	1.6	1.0	2.2								
	LC	1.9	2.0	1.7	1.8			2.4	2.5	361	891	8	20	16	39	3.6	9
Alachlor	WP	3.2	7.1			3.8	8.3	2.6	5.6								
	LC	1.5	1.6			1.6	1.7	1.6	1.7	397	981			83	206	60	148
Propachlor	WP	1.1	2.4					0.91	2.0								
	GP	3.4	7.5			4.0	8.7	3.0	6.6	1,056	2,609			783	1,934	4.0	10
2,4-D ester	LC	0.4	0.4	0.6						89	221	13	33				
EPTC and 2,4-D	GP	2.9	6.4							33	82						
CORN - POSTEMERGENCE																	
All herbicides	WP	1.0	2.2	0.96	2.1	0.87	1.9	0.87	1.9								
	GP					3.4	7.4										
	LC	0.4	0.4	0.5	0.5	0.5	0.5	1.0	1.1	1,382	3,415	395	975	1,523	3,764	431	1,065
Atrazine	WP	0.96	2.1	1.0	2.2	0.73	1.6	0.87	1.9	133	328	132	326	102	253	221	546
Atrazine and oil	WP	1.0	2.2	0.96	2.1	0.91	2.0	0.91	2.0	145	359	161	398	679	1,678	147	362
2,4-D amine	LC	0.4	0.4	0.5	0.5	0.5	0.5	0.6	0.6	442	1,093	45	112	446	1,103	36	88
2,4-D ester	LC	0.4	0.4	0.5	0.5	0.4	0.4	0.7	0.7	517	1,277	31	77	201	497	15	39
SOYBEANS																	
All herbicides	WP	0.82	1.8	0.78	1.7	0.82	1.8	0.82	1.8								
	GP	3.5	7.7	3.7	8.1	4.1	9.0	3.6	7.8								
	LC	1.2	1.3	1.6	1.7	1.1	1.2	1.5	1.6	2,286	5,648	153	378	792	1,598	23	58
Alachlor	GP	3.6	7.9	3.1	6.8	4.1	9.0	2.3	5.0								
	LC	1.4	1.5	1.7	1.8	1.3	1.4	2.3	2.4	336	830	7	18	115	283	4.5	11
Amiben	GP	3.5	7.7	3.7	8.2	4.1	9.0	3.8	8.4								
	LC	2.1	2.2	2.0	2.1	1.7	1.8	2.3	2.4	1,097	2,711	38	95	457	1,128	5.3	13
Linuron	WP	0.59	1.3	0.78	1.7	0.87	1.9	1.0	2.2	130	322	70	173	6	14	4.5	11
Trifluralin	LC	0.9	0.9	1.0	1.1	0.9	0.9	1.0	1.0	496	1,226	4	11	186	460	3.2	8
Alachlor and linuron	WP	0.50	1.1	0.64	1.4			0.50	1.1								
	LC	1.2	1.3	1.0				1.0	1.1	94	232	22	54			5.3	13
SMALL GRAINS																	
All herbicides	LC	0.5	0.5	0.5	0.5	0.4	0.4	0.5	0.5	38	93	119	295	1,562	3,859	159	393
2,4-D amine	LC	0.5	0.5	0.5	0.5	0.4	0.4	0.6	0.6	27	68	43	107	664	1,641	71	176
2,4-D ester	LC			0.5	0.5	0.4	0.4	0.6	0.6			28	70	89	221	32	78
MCPA	LC	0.5	0.5	0.5	0.5	0.4	0.4	0.5	0.5	6	15	43	106	726	1,794	54	135

*Before diluting, WP is weight of wettable powder; GP is weight of granular powder; LC is volume of liquid concentrate

** Where no rates are shown, no or insufficient reports were received to obtain conclusive data.

* Areas treated more than once are counted for each treatment.

TABLE 9

Rate of application and estimated area treated with major insecticides in four Great Lakes States for 1971. (Recalculated from General Farm Use of Insecticides 1971 Wisconsin and Illinois, Michigan and Minnesota, Wisconsin Statistical Reporting Service, Sponsored by the Wisconsin Department of Agriculture and U.S. Department of Agriculture (205)).

Insecticide	Formulation*	Average rate of application per unit area**								Estimated area treated*							
		Illinois		Michigan		Minnesota		Wisconsin		Illinois		Michigan		Minnesota		Wisconsin	
		kg or liter	lb or qt	kg or liter	lb or qt	kg or liter	lb or qt	kg or liter	lb or qt	Ha x10^3	A x10^3	Ha x10^3	A x10^3	Ha x10^3	A x10^3	Ha x10^3	A x10^3
CORN - PREEMERGENCE																	
All insecticides	WP	1.8	4.0					0.82	1.8								
	GP	2.9	6.3	2.2	4.9	3.3	7.3	3.1	6.9	2,221	5,488	110	271	673	1,663	338	835
	LC	1.1	1.2			0.57	0.6	0.76	0.8								
Aldrin	GP	2.5	5.5	2.0	4.3	2.3	5.1										
	LC	1.2	1.3			0.85	0.9			915	2,261	67	166	64	158		
Buxten	GP	3.4	7.4	3.5	7.7	3.8	8.4	3.2	7.0	224	554	5.7	14	138	341	32	79
Diazinon	GP	2.8	6.1	3.4	7.4	3.2	7.1	2.0	4.4	93	230	15	38	38	94	27	67
Heptachlor	GP	2.6	5.8			1.1	2.5			173	428	13	31	12	30		
Phorate	GP	3.0	6.6			3.0	6.6	2.8	6.2	376	928			72	179	81	201
Carbofuran	GP	3.7	8.1			3.5	7.8	3.5	7.7	289	713			240	593	172	424
CORN - POSTEMERGENCE																	
All insecticides	WP					1.6	3.5	0.73	1.6								
	GP	2.8	6.1			3.2	7.0	2.7	5.9	90	223	1.2	3	55	136	30	75
	LC					0.85	0.9										
Diazinon	GP					3.2	7.0	3.0	6.7					38	94	12	30
Phorate	GP	2.9	6.5					2.3	5.0	36	90					8.9	22
SOYBEANS																	
All insecticides	LC					1.4	1.5			52	129	1.6	4	4.9	12		
SMALL GRAINS																	
All insecticides	WP			0.64	1.4												
	LC			1.0	1.1			0.76	0.8	20	49	85	209	1.6	4	2.8	7
Carbaryl	WP			0.64	1.4							70	172				
Malathion	LC			1.0	1.1			0.76	0.8			8.5	21			2.4	6
HAY																	
All insecticides	WP			0.78	1.7			0.45	1.0								
	LC	1.5	1.6	1.3	1.4			1.2	1.3	21	51	68	168	1.6	4	3.2	8
Azinphosmethyl	LC			0.85	0.9							16	39				
Carbaryl	WP			0.78	1.7							19	48			1.2	3
Diazinon and methoxychlor	LC	2.0	2.1	1.5	1.6					12	30	10	25				
Malathion	LC			1.3	1.4							6.9	17				
Malathion and methoxychlor	LC			1.9	2.0							12	30			1.2	3

*Before diluting, WP is weight of wettable powder; GP is weight of granular powder; LC is volume of liquid concentrate

** Where no rates are shown, no or insufficient reports were received to obtain conclusive data.

*Areas treated more than once are counted for each treatment.

TABLE 10

Insecticides used in processing crops, fruit and vegetable production for the Lake Michigan region of Wisconsin in 1962 and 1969. Excerpted and recalculated from Libby, Wisconsin Commercial Fruit and Vegetable Production Crop Acreage and Insecticide Use Survey, Mimeo Rept. (107), Dept. of Entomology, Univ. of Wisconsin, College of Agriculture, and Libby and Koval, Wisconsin Commercial Fruit, Vegetable, and Processing Crop Acreage and Insecticide Use Survey, Resource Rept. 13 (108), University of Wisconsin-Extension.

Insecticide	1962				1969			
	Area treated*		Insecticide used		Area treated		Insecticide used	
	Ha	A	kg	lb	Ha	A	kg	lb
Aldrin	4,484	11,079	8,602	18,960	0	0	0	0
Azinphosmethyl	138	341	870	1,918	3,018	7,458	10,880	23,980
Bacillus thuringiensis	--	--	--	--	369	911	--	--
Carbaryl	1,015	2,508	16,175	35,651	4,871	12,037	38,503	84,864
Chlordane	20	50	38	83	223	550	628	1,385
Chloropropylate	--	--	--	--	53	131	150	331
DDT	8,266	20,426	110,143	242,768	1,727	4,267	9,095	20,046
Diazinon	190	470	843	1,857	1,028	2,541	9,343	20,592
Dieldrin	1,979	4,890	3,549	8,043	22	54	31	68
Demeton	18	45	10	23	240	594	79	175
Dimethoate	--	--	-	--	84	207	47	104
Disulfoton	--	--	--	--	1,710	4,226	4,566	10,063
Endosulfan	559	1,381	865	1,907	2,328	5,753	4,445	9,797
Endrin	1,744	4,310	916	2,019	0	0	0	0
Ethion	380	939	796	1,754	357	881	511	1,126
Gardona	--	--	--	--	225	557	2,594	5,718
Genite	69	170	309	680	0	0	0	0
Heptachlor	3.2	8	1.8	4	0	0	0	0
Kelthane	127	315	129	284	74	182	142	312
Lead arsenate	1,256	3,104	48,082	105,978	1,828	4,516	39,407	86,858
Malathion	2,452	6,059	26,239	57,834	2,077	5,133	13,738	30,280
Methoxychlor	798	1,971	20,774	45,788	96	236	1,102	2,429
Meta-SystoxR	--	--	--	--	47	115	26	58
Mevinphos	956	2,363	686	1,512	385	952	282	622
Methomyl	--	--	--	--	1,197	2,957	3,039	6,698
Mitox	49	122	146	322	0	0	0	0
Morestan	--	--	--	--	34	83	44	96
Naled	269	665	302	665	156	385	250	550
Ovex	191	473	344	758	0	0	0	0
Parathion	4,340	10,725	17,169	37,843	5,065	12,515	11,576	25,514
Phorate	2,013	4,975	5,876	12,952	5,688	14,055	16,283	35,890
Phosphamidon	40	100	224	494	243	600	1,250	2,755
Perthane	0	0	0	0	25	61	86	189
Rotenone	0	0	0	0	1.2	3	2.3	5
TDE (DDD)	475	1,174	6,211	13,690	89	221	714	1,573
TEPP	42	105	44	98	26	64	188	415
Tetradifon	218	538	461	1,105	74	183	126	277
Toxaphene	2,955	7,302	34,732	76,553	1,040	2,570	7,609	16,770
Trithion	43	106	37	81	5.3	13	7.3	16
VC-13	451	1,114	360	794	257	636	342	754
TOTAL	35,544	87,829	305,036	672,331	34,661	85,647	177,086	390,310

*Door County not included

A comparison between the survey in 1962 (107) and that in 1969 (108) revealed a decreasing trend in the area treated and total quantity of insecticide applied (Table 10). However, there was a significant shift in the kinds of chemicals used. The use of organochlorine insecticides, especially DDT, aldrin, methoxychlor, TDE (DDD), toxaphene, dieldrin and endrin,was either decreased drastically or discontinued entirely. This was accompanied by a rapid increase in the use of carbamate (particularly carbaryl) and organophosphorous (diazinon, phorate, disulfoton, and phosphamidon) insecticides.

It has been shown in the preceding discussion that pesticides are used widely in the Great Lakes Region. In one way or another, more than half of the harvested area for crop production has been treated with pesticides. Herbicides are the predominant pesticides used and their use has been increasing in recent years. Shifts to use of different insecticides are evident. The use of organophosphorous and carbamate insecticides has increased, while the use of organochlorine insecticides has declined sharply. Field crops receive most of the pesticides, particularly the herbicides and insecticides. Major proportions of the herbicides and insecticides have been used to protect row crops, i.e., corn and soybeans. Sweet corn, potatoes and apples received considerable insecticidal treatment.

The surveys mentioned, though not accurate, represent the best available data. Obviously, more comprehensive inventories of the kinds and amounts of pesticides used in the region are needed, especially since some organochlorine insecticides were placed on the restricted use list.

3.2 Length of Time of Pesticide Usage Practice on Agricultural Land

The use of pesticides dates back a hundred years, but usage did not increase tremendously until about 1945 when DDT was introduced commercially. A rapid growth of the organic pesticide industry followed for the next two and a half decades (184). Chemical control has revolutionized agricultural production, with the attainment of high crop yields never before realized. Almost all agricultural practices formerly used to combat weeds, insects, and diseases were shifted to favor the use of pesticides. Since the intro-

duction of DDT, a myriad of chemicals have been synthesized. Approximately 34,500 pesticide products are currently registered with EPA and USDA (67). Farmers use the largest volume of pesticides, but about one half of the registered products are utilized by persons other than farmers.

DDT was believed to be used first in the Great Lakes watershed in the late 1940's (148). Although no comprehensive record exists of the types and amounts of pesticides used for various crops grown in the watershed, fragmentary evidence presented under Section 3.1.2 shows that pesticides have been used extensively, probably for the past 25 years. Types and amounts of pesticides used depend upon the crop grown. Field crops and orchards have been treated quite heavily while pasture land received little. Only in recent years have attempts been made to restrict the use of the persistent organochlorine insecticides and replace them with the less persistent organophosphorous and carbamate types of insecticides.

3.3 Types and Nature of Pesticide Pollutants

All agriculturally-applied pesticides are potential pollutants of aquatic ecosystems. The type and nature of pesticide residues reaching the Great Lakes from agricultural watersheds is controlled largely by: 1. amounts, intensities, and lengths of time pesticides have been applied; 2. persistence and/or residence time in the watershed; and 3. mobility of residues.

3.3.1 Insecticides

The organochlorine insecticides are the pesticides most studied from the standpoint of environmental contamination. Because of their long persistence in soils they provide a greater potential than other pesticides for contaminating aquatic systems. Furthermore, their low water solubilities make them strongly adsorbed to soil colloids, facilitating transportation by erosion. Numerous persistency studies indicate that the residence time of such commonly-used compounds as DDT, dieldrin, aldrin, chlordane, heptachlor, and lindane in soils is several years (45), and DDT is detected

frequently over a decade after application.

Use of organophosphorous and carbamate insecticides increased rapidly following restriction on the use of organochlorine insecticides in many Great Lakes States after 1969. Although organophosphorous compounds are more toxic to mammals than the organochlorine compounds, they persist for a relatively shorter length of time in soils (94). Similarly, the carbamates do not persist long in soils (158).

3.3.2 Herbicides

Herbicides are generally less ubiquitous in the environment than organochlorine insecticides. However, compounds such as s-triazine herbicides, picloram, monuron and related substituted ureas, and 2,4,5-T often remain in soils for at least a year after application (5). Atrazine is a widely-used herbicide for field crops, particularly corn. 2,4-D and the carbamate herbicides are short-lived in soils.

3.3.3 Fungicides

Unlike herbicides and insecticides, the fate of fungicides in soils is practically unknown. Major usage of fungicides occurs in orchards and vegetable farms, which abound in the Great Lakes watersheds. Mercurial fungicides have been a recent concern following the detection of mercury contamination of the Great Lakes in early 1970 (182). Since the use of mercury fungicides has been severely restricted, agricultural usage of mercury is negligible in comparison to industrial usage (154). In addition, the proportion of croplands treated with fungicides is small compared to the area treated with herbicides and insecticides. Considering these facts, it is doubtful that agricultural fungicides are contributing pollution to the Great Lakes.

3.3.4 Degradation Products

It is well known that the metabolites detected usually in the partial

degradation of DDT are DDE in aerobic systems and TDE (DDD) in anaerobic systems. DDE appears to be resistant to aerobic and anaerobic metabolism, thus its accumulation is possible in aquatic environments. Other pesticides may be metabolized to compounds equally or more active than the parent compound, as shown in the epoxidation of aldrin and dieldrin. Metabolites of many pesticides have not, as yet, been clearly identified, nor have their possible toxicity, persistence and accumulation in the aquatic ecosystem been defined adequately.

3.4 State-of-the-Art in Assessing and Quantifying Problems

The discussion of this section is covered extensively and is integrated in several sections, namely, 3.1.2, 3.5, and 3.6, and no further discussion is presented in this section.

3.5 Mobility of Pesticides in Soil, Groundwater, and Surface-Water Systems

Pesticides used to protect crops and pastures from weeds, insects, and diseases may enter the Great Lakes via: 1. the atmosphere, 2. groundwater, and 3. surface runoff. The discussion on pesticide entry will not be limited to information gathered in and around the Great Lakes Basin, but will include pertinent information obtained elsewhere.

A general scheme of the distribution and fate of herbicides in the biosphere is shown in Fig. 1 (55). Although the diagram was prepared for herbicides, it is equally applicable to other pesticides and is entirely self-explanatory.

3.5.1 Pesticide Movement to the Atmosphere

The atmosphere becomes contaminated with pesticides by: 1. drift during application, 2. volatilization, direct or from soil and other surfaces, and 3. wind erosion. That the atmosphere is contaminated with pesticides is well-documented (52).

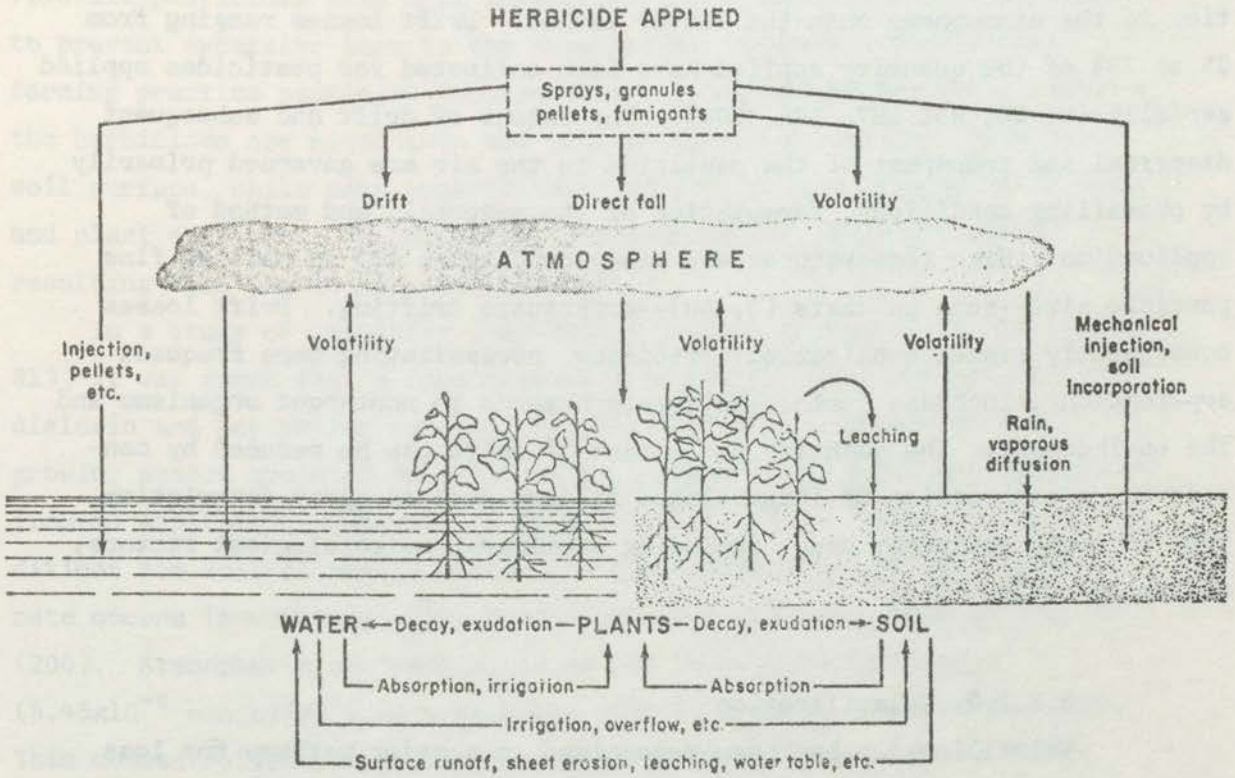

Fig. 1 Scheme showing the possible distribution and fate of herbicides and their degradation products in the biosphere. Redrawn from Foy and Bingham (55).

3.5.1.1 Drift

Drift is that portion of spray that is moved away from the target area by wind or other meteorological factors. The importance of inadvertent losses from drifting sprays and dusts has been fully recognized (130). Aerial spraying, which is a more widely used method of application as opposed to ground application (124, 184), contributed more pesticide pollution to the atmosphere than the latter method. Drift losses ranging from 25 to 75% of the quantity applied have been estimated for pesticides applied aerially (4, 28, 85, 187, 188, 206). The extent of drift and subsequent dispersal and transport of the pesticide in the air are governed primarily by prevailing conditions, formulation of the material, and method of application. High temperatures and windy conditions (3) as well as fine particle size--such as dusts (3, 60)--accentuate drifting. Drift losses consequently reduce application efficiency, necessitating more frequent applications, increase costs, and create hazards to nontarget organisms and the environment. The quantity and extent of drift can be reduced by considering a combination of interrelated factors such as spray formulation, type of spray equipment used, including atomizers, meteorological factors, and spray method (118).

3.5.1.2 Volatilization

Volatilization has been recognized as a major pathway for loss of pesticides from soil, plant and water surfaces. The factors and mechanisms of volatilization losses in soils are discussed in a comprehensive review (177).

After volatilization related to application has occurred, the remaining pesticides ultimately reach the soil. In soil, loss is evident for both surface-applied and soil-incorporated pesticides. The vaporization rate of a pesticide is related to its vapor pressure, but once it is in contact with the soil its vapor pressure is modified by environmental variables. Climatic and edaphic factors regulating the volatilization rate include: air movement (29, 51, 78), temperature (51, 70, 78, 173, 202), relative humidity of the surrounding air (11, 87), soil moisture content (78, 87, 97,

142, 173, 175, 176, 200, 202), soil organic matter content (2, 70, 143, 173) and pesticide concentration in the soil (51, 175).

Field measurements indicate that significant volatilization loss may occur if pesticides are not incorporated in the soil. Losses have been observed for DDT (85, 200), TDE (DDD) (200), dieldrin (30, 176, 202), endrin (201), TDE (200), lindane (174, 175, 176), heptachlor (31), and IPC and CIPC (142). It appears that incorporation of some of the more volatile pesticides into soil immediately after application is necessary to prevent excessive loss to the atmosphere. However, under normal farming practice pesticide incorporation is not always possible. Many of the herbicides are mixed into the soil or banded in the row below the soil surface, while many insecticides and fungicides are applied on soil and plant surfaces. The potential for atmospheric pollution exists, resulting from pesticide volatilization.

In a study of pesticide loss from a watershed cropped to corn (30, 31), it was found that a considerable fraction of soil-incorporated dieldrin and heptachlor was lost by volatilization. Losses in one growing season amounted to 2.8-2.9% for dieldrin and 3.9% for heptachlor from an application rate of 5.6 kg/ha (5.0 lb/acre). Under field conditions the volatilization process is continuous although the highest rate occurs immediately after pesticide application as shown in Fig. 2 (200). Atmospheric concentrations of DDT as high as 2,040 ng/m^3 (5.45×10^{-8} $ounces/yd^3$) were measured initially above a nonflooded plot. This concentration dropped to 100 ng/m^3 (2.67×10^{-9} $ounces/yd^3$) 2 days after application. Corresponding levels of TDE (DDD) decreased from 575 to 92 ng/m^3 (1.54×10^{-8} to 2.45×10^{-9} $ounces/yd^3$) within the first 2 days. Except for the first few days after application, pesticide concentrations seldom exceeded 100 ng/m^3 (2.67×10^{-9} $ounces/yd^3$), and subsequent change in atmospheric concentrations during a 6-month period appeared to be related to climatological factors.

3.5.1.3 Wind Erosion

Pesticides dispersed in the atmosphere become associated with airborne particulate matter and, as such, are capable of being transported

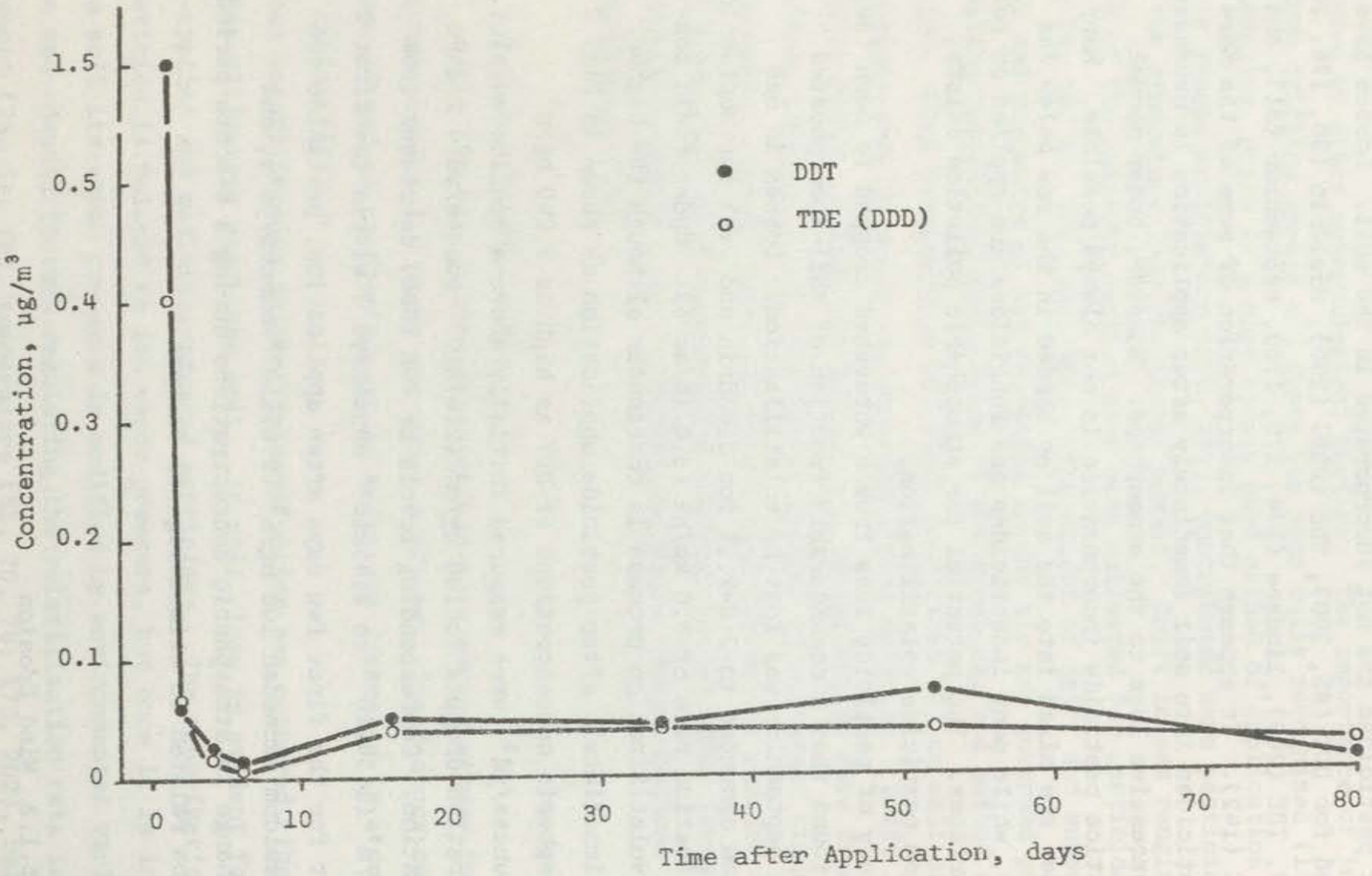

Fig. 2. Atmospheric concentration of DDT and TDE (DDD) monitored at 30 cm above the soil surface. Figure assembled from data of Willis et al. (200). $\mu g/m^3 = 2.67 \times 10^{-2}$ ounces/yd^3

(38, 194). Their distribution away from application sites is dependent on the prevailing wind movement and rate of fallout. Pesticide-laden dust may also originate from dislodgement of the soil-pesticide complex due to wind erosion (38, 121). Wind erosion may provide an important atmospheric source of pesticides when the pesticides are redeposited on watersheds and waterbodies.

Earlier discussions show ample evidence of atmospheric contamination by pesticides from drifting spray, by air-borne particulate matter, and through volatilization from soil surfaces. By these means, the potential for polluting areas not treated directly with pesticides is great when the pesticides are removed from the atmosphere through gravitational fallout and rain washout. Precipitated dust collected in Cincinnati after a dust storm indicated the presence of DDT, chlordane, Ronnel, and DDE as the major pesticide components of the dust (38). Heptachlor epoxide, 2,4,5-T and dieldrin were present in lesser amounts. The concentration ranged from 3 ng/g (3 ppb) dieldrin to 600 ng/g (600 ppb) DDT based on the air-dried weight of the dust. It was believed that the dust originated in the southeast of the United States where agricultural fields were treated heavily with pesticides. A country-wide attempt to determine atmospheric contamination by pesticides was performed by collecting samples in nine locations which included urban and rural sampling sites (178). Nineteen pesticides and metabolites were sought in the samples. Detected pesticide levels ranged from the lower level of detection of 0.1 ng/m^3 air (2.67×10^{-12} ounces/yd^3) to as high as 1,560 ng/m^3 (4.16×10^{-8} ounces/yd^3) *p,p'* DDT, 2,520 ng/m^3 (6.73×10^{-8} ounces/yd^3) toxaphene, and 465 ng/m^3 (1.24×10^{-8} ounces/yd^3) parathion. Only DDT was found in all localities. Levels of DDT in the atmosphere were higher in agricultural areas, and atmospheric content was more closely correlated with spraying activities than with rainfall.

Data revealed that DDT distribution in coastal and oceanic waters results from fallout of airborne particulate material (151). This implies that a vast area of water surface, such as the Great Lakes, may receive significant inputs of pesticides from the atmosphere (89). Normal levels of airborne pesticides could be low, but the return of these compounds

either in solution or in the adsorbed state to the earth's surface is continuous; however, their contribution to aquatic systems is difficult to evaluate (144). These compounds, if dissolved by rainfall, may gain entry to the aquatic system or be deposited on watersheds and become adsorbed by soil colloids and suspended materials.

The presence of pesticides in the atmosphere is well-documented. However, little is known about the ultimate fate of these compounds after they enter the atmosphere. More research is needed to evaluate the photochemical alteration, adsorption-desorption on dust particles, fallout and rainout of airborne pesticides.

3.5.2 Pesticide Movement to Groundwater

Pesticide contamination of groundwater can occur through leaching. Downward movement of agriculturally-applied pesticides is controlled by soil, pesticide, and climatic factors. These controlling factors are well-documented in reviews (1, 10, 82, 179, 190). Leachability of a compound depends primarily on the degree to which it is adsorbed to soil colloids. Adsorption is associated closely with organic matter content and occasionally with clay for nearly all pesticides. Pesticides are leached more readily in coarse-textured than in fine-textured soils as the latter contain more clay and generally more organic matter. Furthermore, water infiltration is relatively faster in coarser-textured soils. The solubility of pesticides plays an important role in their movement in the soil since solubility limits the concentration of the compound in the soil-water phase. Solubility--which is largely dependent on the chemical constitution of the compound--is shown to be correlated inversely with adsorptivity. Thus, pesticides of high water solubility are more subject to vertical movement than those of low water solubility. The transport of pesticides through the soil is conditioned by the amount, intensity and frequency of percolating water. Water facilitates desorption as well as dissolution of particulate or adsorbed compounds.

Adsorption of nonionic pesticides--which include the organochlorine and organophosphorous insecticides--is correlated primarily with soil organic matter content (2, 10, 16, 64, 68, 76, 80, 81, 86, 101, 143, 150)

and to a lesser extent with clay content. Retention of acidic and basic compounds is affected markedly by soil pH (44, 75, 79, 189, 192). Soil reaction controls the overall charge of the molecule and hence its adsorptivity to clay and organic colloids. The organic cations, diquat and paraquat, are held strongly by clay minerals and are often adsorbed irreversibly (193). Weakly-adsorbed water-soluble compounds are desorbed readily by water and hence pose a greater potential for leaching, depending upon the composition and amounts of the soil colloidal fractions.

Numerous studies have been conducted to determine the relative mobilities of pesticides in soils. The findings of these investigations are summarized in Table 11 (82). The organochlorine insecticides--which have limited water solubility--are the least mobile, followed by the organophosphorous insecticides. The water-soluble acidic herbicides are most mobile. Most of the pesticides, such as triazines, phenylureas and carbamates, have intermediate mobility. Within a diverse group of pesticides, relative mobility is related essentially to solubility.

Organochlorine insecticides are, in general, nonleachable. Field trials have shown that they are retained largely in the upper 15 to 20 cm layer of most agricultural soils (14, 33, 37, 46, 102, 112, 117, 133, 167, 180, 199, 203) including aquifers (159). Any movement to lower depths and subsequently to water tables is attributed by some investigators to be the result of the physical transport of adsorbed or particulate compounds by water through vertical cracks formed during dry periods (180, 199). Thus, holes made by soil animals may be important in the downward movement of pesticides.

Limited data on pesticide concentration in groundwater collected on treated watersheds and in wells indicate that low or negligible quantities of pesticides are transported through the soil profile (88, 90). However, in areas with a shallow and fluctuating water table, frequently found in coastal sandy soils (102) and aquifers (159), appreciable amounts of pesticide may find their way to the underlying water; this is especially true for the soluble herbicides. Toxaphene and fluometuron which reached groundwater were found by LaFleur et al. (102) to persist for a year.

In summary, the entry of a particular pesticide to groundwater is defined by soil type, depth to water table, rainfall infiltration and

TABLE 11

Relative mobility of pesticides in soils*
(Adapted from Helling et al. (82).)

Mobility Class**				
5	4	3	2	1
TCA+	Picloram	Propachlor	Siduron	Neburon
Dalapon	Fenac	Fenuron	Bensulide	Chloroxuron
2,3,6-TBA	Pyrichlor	Prometone	Prometryne	DCPA
Tricamba	MCPA	Naptalam	Terbutryn	*Lindane*
Dicamba	Amitrole	2,4,5-T	Propanil	*Phorate*
Chloramben	2,4-D	Terbacil	Diuron	*Parathion*
	Dinoseb	Propham	Linuron	*Disulfoton*
	Bromacil	Fluometuron	Pyrazon	Diquat
		Norea	Molinate	*Chlorphenamidine*
		Diphenamid	EPTC	Dichlormate
		Thionazin	Chlorthiamid	*Ethion*
		Endothall	Dichlobenil	*Zineb*
		Monuron	Vernolate	Nitralin
		Atratone	Pebulate	C-6989
		WL 19805	Chlorpropham	*ACNQ*
		Atrazine	*Azinphosmethyl*	*Morestan*
		Simazine	*Diazinon*	*Isodrin*
		Ipazine		*Benomyl*
		Alachlor		*Dieldrin*
		Ametryne		*Chloroneb*
		Propazine		Paraquat
		Trietazine		Trifluralin
				Benefin
				Heptachlor
				Endrin
				Aldrin
				Chlordane
				Toxaphene
				DDT

*From data of Gray and Weirlich (64), C. I. Harris (77), Helling (81), Koren et al. (101), Nash and Woolson (133), and many other references.

**Class 5 compounds (very mobile) to Class 1 compounds (immobile); in each class pesticides are ranked in estimated decreasing order of mobility.

+Names of herbicides are set in normal type; insecticides, fungicides, and acaricides in *italics*.

persistence of the compound in the soil. Available information indicates that pesticide contamination of groundwater under normal agricultural usage is minimal. However, a need exists for more information on the extent of pesticide pollution as pesticides are known to enter underlying water through cracks and through sand and aquifers. The fate of pesticides in subsurface environments and movement of contaminated groundwater to aquatic systems is absolutely unknown (155).

3.5.3 Pesticide Movement by Surface Runoff

Pesticides present on agricultural land may be transported through surface runoff either in solution and/or as adsorbed molecules on suspended eroded soil particles. Since surface soils are susceptible to erosion, pesticides retained in the plow-depth layer are potentially transportable by surface drainage. The rate and magnitude of pesticide loss by runoff depends principally on soil properties, nature of the pesticide, and climatic factors such as frequency, intensity, and duration of rainfall (9, 22, 49, 50, 84, 138). Modifying factors include topography, vegetative cover, rate of application, time between application and first rainfall, and size of the watershed (12, 15, 24, 73, 199). The movement and persistence of pesticides in the soil affect the amount of residues present in the surface soil as well as that in runoff. Factors affecting the movement and persistence of pesticides in soil have been reviewed (10, 45, 82, 95).

Surface runoff from agricultural watersheds has been implicated as the major avenue of pesticide input to the Great Lakes and other aquatic systems (35, 103, 131, 139, 148, 204), especially the organochlorine insecticides. As yet, no comprehensive effort has been made to determine the contribution of agricultural runoff to the total pesticide burden of the environment. Investigations conducted in small treated watersheds demonstrated the lateral movement of pesticides, and data from several studies are summarized in Table 12 (144). Table 12 reveals that losses of most organochlorine insecticides relative to the amount applied are low even for surface-applied insecticides. Once the soils are contaminated with the more persistent organochlorine insecticides like DDT, these

TABLE 12

Runoff losses of agriculturally applied pesticides under various conditions. (Adapted from Pionke and Chesters (144).)

Pesticides and literature reference ( )	Amount applied, kg/ha	Type of application**	Crop	Soil texture	Avg slope, %	Plot size	Duration of experiment, months	Amount of runoff, cm	Pesticide loss in runoff, % of application	Range of pesticide loss in runoff increments, ug/g	Location
Aldrin (72)	1.5	Ss	Cultivated	sil	1-2	3x1.2 m	++	4.3¶	5.2	##	KY
Dieldrin (30)	5.6	Inc	Primarily corn	sil	14	1.09 ha	26	3.1	0.07	0.4-4.1	Coshocton, OH
Dieldrin (30)	5.6	Ioc	Primarily corn	sil	14	0.68 ha	8	3.1	0.007 2.2	(1.9-20 ug/l water) (1.6-14 ug/g sediment)	Coshocton, OH
Dieldrin (72)	1.5	Ss	Cultivated	sil	1-2	3x1.2 m	++	4.3¶	4.7	##	KY
DDT (72)	1.5	Ss	Cultivated	sil	1-2	3x1.2 m	++	4.3¶	6.3	##	KY
DDT (49)	0.73*	Ss***	Cont. potatoes	gl	8	3.7x21 m	24	9.7	1.6	7.0-8.3	Presque Isle, ME
DDT (49)	0.73*	Ss***	Rotation potatoes	gl	8	3.7x21 m	24	7.9	1.0	Trace-67	Presque Isle, ME
DDT (49)	0.73*	Ss***	Oats	gl	8	3.7x21 m	12	3.6	0.01	Trace-22	Presque Isle, ME
Endosulfan (49)	1.0	Ss***	Cont. potatoes	gl	8	3.7x21 m	12	5.1	0.35	1.0-19	Presque Isle, ME
Endosulfan (49)	1.0	Ss***	Rotation potatoes	gl	8	3.7x21 m	12	4.8	0.25	Trace-18	Presque Isle, ME
Endosulfan (49)	0.72*	Ss***	Oats	gl	8	3.7x21 m	12	3.6	0.01	Trace-3	Presque Isle, ME
Endrin (49)	1.5	Ss***	Cont. potatoes	gl	8	3.7x21 m	24	9.7	0.9	1.0-49	Presque Isle, ME
Endrin (49)	1.5*	Ss***	Rotation potatoes	gl	8	3.7x21 m	24	7.9	0.6	Trace-48	Presque Isle, ME
Endrin (199)	0.337	Sg	Sugarcane	scl	0.2	7.3x61 m	10	1.2§	0.1	<0.01-2.73	Baton Rouge, LA
Endrin (199)	0.337	Sg	Sugarcane	scl	0.2	7.3x61 m	5	0.3	0.2	0.15-5.02	Baton Rouge, LA
Methoxychlor (46)	22.4	Sl	Grass	sil	-	0.07 m²†	14	27	0.04	0.1-8.8	Coshocton, OH
Atrazine (196)	3.0	Ss	Fallow	sl	6.5	1.8x10.7 m	+++	3.9	5.7-7.3	(100-10,340 ug/l water) (0.5-88 ug/g sediment)	Watkinsville, GA
Atrazine (152)¶¶	3.0	Ss	Corn surf. contour.	sil	10-15	0.8-1.5 ha	1	0.4	15	(1,170-4,910 ug/l water) (1.77-7.35 ug/g sed.)	Castana, IA
Atrazine (152)	3.0	Ss	Corn ridged contour.	sil	10-15	0.8-1.5 ha	1	1.5	2.2	--	Castana, IA
Propachlor (152)	6.0	Ss	Corn surf. contour.	sil	10-15	0.8-1.5 ha	1	1.0	2.6	--	Castana, IA
Dicamba (181)	1.0-2.0	Ss	Fallow-sod	cl	3-8	3.05x3.05 m	4	<0.2¶	2	0-4,800	College Station, TX
2,4-D-isooctyl (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	+++	10.2¶	40	1,380	Watkinsville, GA
2,4-D-butyl ether (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	+++	10.2¶	35	640	Watkinsville, GA
2,4-D-amine (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	1 hr	9.4¶	5	140	Watkinsville, GA
2,4-D-amine (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	48 hr	8.9¶	4	110	Watkinsville, GA
2,4-D-amine (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	96 hr	8.1¶	5	180	Watkinsville, GA
Picloram (181)	1.0-2.0	Ss	Fallow-sod	cl	3-8	3.05x3.05 m	4	<0.2¶	2.8-5.0	15-5,060	College Station, TX
2,4,5-T (46)	11.2	Sl	Grass	sil	-	0.07 m²†	14	27	0.05	1-380	Coshocton, OH
2,4,5-T (181)	1.0-2.0	Ss	Fallow-sod	cl	3-8	3.05x3.05 m	4	<0.2¶	2	7-3,300	College Station, TX

*Includes residual insecticide present

**Ss = surface spray; Inc. = incorporated 7.5 cm; Sg = surface, granular; Sl = surface, liquid

***Mixed by subsequent cultivation if used on row crops.

†Lysimeter study

††One rainfall occurrence

†††Application 1 to 96 hr before simulated rainfall

¶Generated by simulated rainfall

§Generated, in part, by irrigation

##Combined sediment-water allowed to equilibrate before analysis.

¶¶Addition to Pionke and Chesters (1973) data.

NOTE: kg/ha = 0.892 lb/A; m = 3.282 ft; ha = 2.471 A; m^2 = 10.771 ft^2; cm = 0.394 in; ug/g = ppm.

compounds will reside in the soil for an indefinite period of time and are capable of being carried from one season to the next (84). Thus, the possibility exists that persistent pesticides associated with the soil are subjected continually to runoff, thereby providing a steady, low-level residue to aquatic systems.

Generally, investigations show that pesticide concentrations in runoff are many times greater in the period immediately following application than at later times (12, 22, 30, 46, 49, 152, 181, 199). This period may last for as much as 4 months but is time-dependent. Levels at later periods would depend mainly on the pesticide reservoir residing in the soil that is available for transport. Processes such as volatilization, adsorption, degradation, photodecomposition, and leaching will determine, to a large extent, the level of residues remaining at a particular time.

Periods of accelerated loss following application are observed for several herbicides (Table 12). Rainfall or irrigation occurring immediately following application caused large losses of weakly adsorbed compounds (12, 181, 196) and accelerated loss of the more strongly adsorbed compounds (30, 49, 72, 84, 199). Losses of 2,4-D varied according to the formulation and decreased with increase in solubility. This suggests that downward movement of more soluble compounds into the soil protects them from runoff.

In most treated farmlands, loss following application appears to be affected not only by the availability of runoff-causing rainfall but also on the elapsed time between application and rainfall. Some studies show that concentration of endrin and aldrin in runoff decreased by a factor greater than two where rainfall was delayed 3 to 7 days (72, 199). The time interval from application to the first irrigation affected the amount of picloram removed with surface water or leached into the soil profile of semi-arid rangelands (157). The lower levels associated with delayed rainfall or irrigation is probably attributable to reduced quantities of residues resulting from degradation, volatilization, and photodecomposition. This is of greater importance with the more volatile and less persistent pesticides. Persistent pesticides tend to accumulate in the top soil making them accessible for transportation during periods of runoff.

It is evident that pesticide concentrations are generally higher in the sediment than in the runoff water (24, 30, 72, 196). This relative distribution between the two phases expressed as ratios of pesticide concentrations in runoff water to that adsorbed to the suspended sediment ranged from 1:2 to 1:1000 for most organochlorine insecticides (30, 72, 150, 168) and 1:10 to 1:20 for triazines (74, 196). Although concentrations were higher in the sediment, greater total losses were associated with the runoff water due to its greater volume. This emphasizes the importance of reducing erosion by sound soil management and conservation practices to minimize the pesticidal load of the runoff.

The concentrations and losses reported relate to runoff as it immediately leaves a specific watershed area. Under actual field conditions, agricultural watersheds are seldom treated completely. Studies have shown that losses of diuron, picloram, 2,4-D, and 2,4,5-T diminished as a function of distance from the treated field (50, 117, 181). All herbicide concentrations dropped below the limit of detection within a few hundred meters below the sprayed area (50). These data suggest that runoff originating from treated areas is diluted by runoff arising from untreated areas. Furthermore, uncontaminated suspended particles may adsorb the pesticide present in the water and, together with the originally contaminated sediment, become partially redeposited across the path of the runoff. The amount of a pesticide entering tributaries and lakes depends upon the reactions it undergoes during its travel over land.

Limited studies have been conducted for monitoring pesticide movement over large agricultural watersheds. In such investigations the drainage streams and/or near-shores of the lakes were sampled usually to determine pesticidal pollution from surrounding drainage areas. In Lubbock County, Texas, 18 rural lakes draining extensively-farmed areas were sampled on a routine basis following runoff-producing rainfall for a period of 18 months (195). No measurable concentrations of any of the herbicides or insecticides used commonly in the area were detected in the water samples. Dieldrin, aldrin, and DDT were the only insecticides found in the lake sediments, with dieldrin being present in almost 80% of the samples. The

sediment samples contained no detectable herbicides.

Seasonal variations in residues of organochlorine pesticides in the water of the Utah Lake drainage system were determined in 1970-71 (25). Definite surges of pesticide [1 µg/l (1 ppb) or more] enter Utah Lake three times a year, i.e., early spring, late spring, and fall, generally corresponding to the application times of pesticides by farmers in the area. The pesticides involved were mainly aldrin and BHC in early spring; heptachlor including heptachlor epoxide and methoxychlor in the late spring; and aldrin, heptachlor and methoxychlor in the late fall.

Runoff losses of DDT and dieldrin occurring in a 40,500 ha (100,000 acres) watershed draining the tobacco belt of southern Ontario to Lake Erie were determined by Frank et al. (58). It was calculated that 10 kg (22 lb) of DDT and 0.6 kg (1.3 lb) of dieldrin reached Lake Erie from the watersheds in runoff water or on suspended material. This represented 0.003% of DDT and 0.004% of the dieldrin resident in the watershed soil which were 325,000 kg (716,300 lb) and 14,500 kg (32,000 lb), respectively. As water and sediment entered Long Point Bay and moved into Lake Erie, insecticide residues became considerably diluted.

In October 1968, chlordane and dieldrin were applied on 1,710 ha (4,225 acres) of land bordering Lake Michigan in southeastern Michigan (Chikaming Township, Berrien County) to control Japanese beetle infestation. The aerial application was made using 2,824 kg (6,225 lb) of technical dieldrin and 5,043 kg (11,115 lb) of technical chlordane. The impact of this application to Lake Michigan was monitored by residue measurements in water, sediment, and mussel tissue before and after treatment at two control and four test stream stations (204). Prior to treatment, mean concentrations of chlordane were <0.2 µg/l (<0.2 ppb) in the water, <50 ng/g (<50 ppb) in the sediment, and 34 ng/g (34 ppb) in mussel tissue. The mean dieldrin concentrations before treatment were <0.02 µg/l (<0.02 ppb) in water, <5 ng/g (<5 ppb) in sediment, and 7.83 ng/g (7.83 ppb) in mussel tissue. Following treatment, the chlordane concentrations in water, sediment and mussel tissue reached individual station highs of 3.4 µg/l (3.4 ppb), 22,000 ng/g (22 ppm), and 7,530 ng/g (7.53 ppm), respectively; and dieldrin

concentrations reached 2 µg/l (2 ppb), 2,000 ng/g (2 ppm) and 1,137 ng/g (1.137 ppm), respectively.

Movement of pesticides from treated fields to stream waters occurred in the first 3 months after application as shown for dieldrin (Fig. 3). Based on the discharge flow to the streams an estimated 5.1 kg (11.3 lb) of dieldrin was contributed to Lake Michigan by stream water in the 21 months following treatment; this represented 0.18% of the total application of 2,824 kg (6,225 lb) dieldrin.

Stream sediments continued to show low, but detectable concentrations of chlordane 12 months after treatment. No further monitoring was done after this period. Dieldrin was still present in the sediment 21 months after application at concentrations of 60 to 250 ng/g (60 to 250 ppb). Chlordane and dieldrin concentrations in the stream sediments were high during periods of high flow. This is indicative of the strong affinity of these pesticides to the eroded soil particles and/or to suspended sediments in the streams. Probably bedload, allochthonous, and autochthonous organic materials play significant roles in the transport and accumulation of insecticides in the Great Lakes.

To determine whether North Shore streams in the Minnesota drainage basin of Lake Superior are important contributors of pesticides, samples of water and clams collected from 24 streams were analyzed for *p,p'*-DDT, *o,p'*-DDT and DDE within an 8-month period starting in May, 1972 (127). Under conditions of normal flow, very little insecticide contamination was being contributed to Lake Superior by those streams. Highest values [20 to 21.8 µg/l (20 to 21.8 ppb) *p,p'*-DDT] occurred in water samples from Lester and Lower St. Louis Rivers following a heavy runoff in October. Ordinarily, median levels in these rivers were 0.03 and 0.02 µg/l (0.03 and 0.02 ppb), respectively. The median value for all other streams was 0.01 µg/l (0.01 ppb). Native clams from the St. Louis River had accumulated appreciable amounts of *p,p'*-DDT ranging from 93 to 1,630 µg/l (93 to 1,630 ppb) indicating that this river carries an appreciable load of DDT on either an intermittent high level during heavy rains or a constant low level basis. Although this study did not isolate the sources of contami-

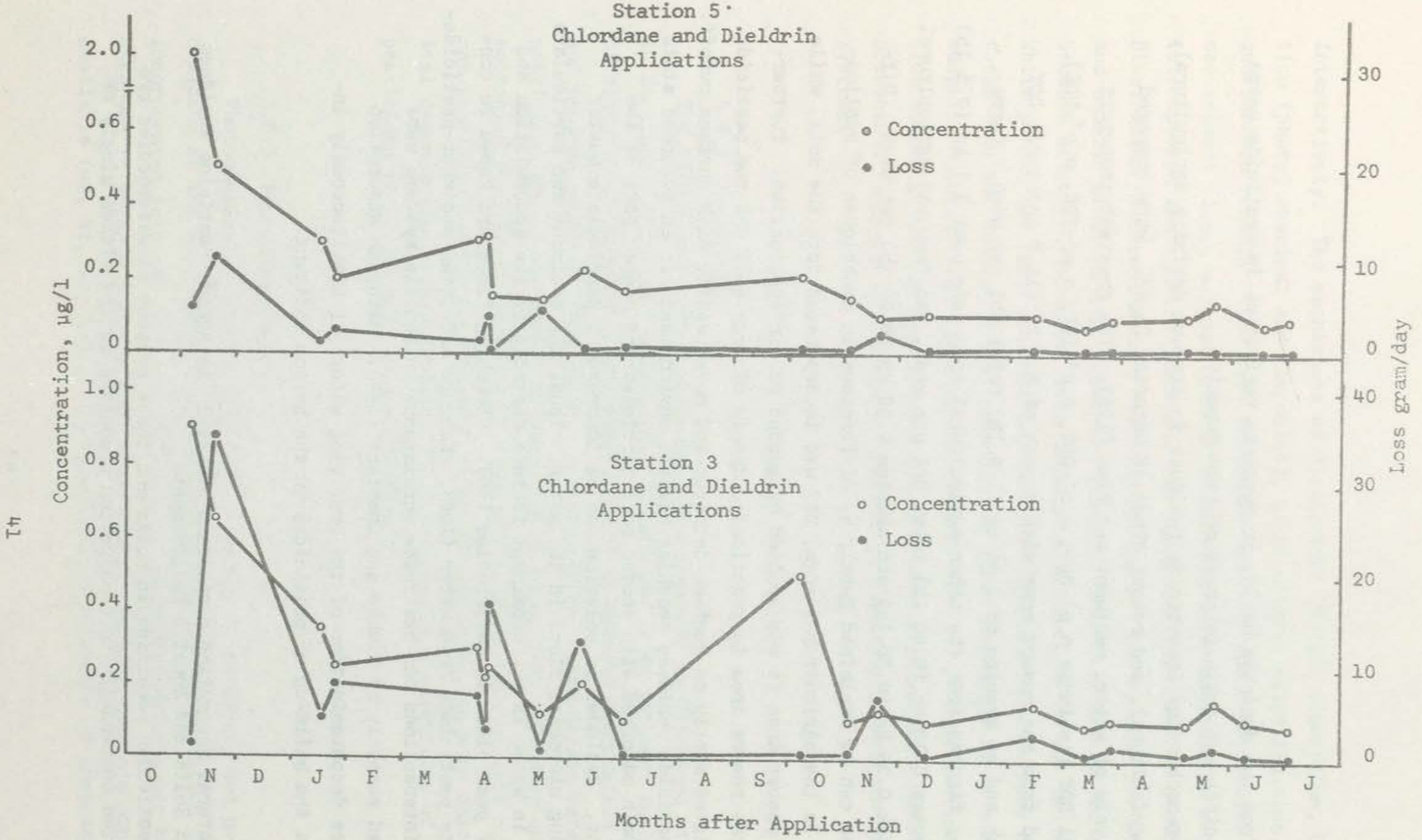

Fig. 3. Concentrations and losses of dieldrin to Berrien County (Michigan) streams, October 1968-July 1970. Modified from Willson et al. (204). µg/l=ppt, gram/day = 0.035 ounces/day.

nants from the drainage basin, it suggests variations in pesticide usage in the different drainage areas of Lake Superior.

Organochlorine insecticide residues in streams draining agricultural, urban-agricultural, and resort areas of Ontario, Canada, were compared by analysis of water, sediment and fish (125). The greatest transport of total DDT (includes *p,p'*-DDT, *o,p'*-DDT, *p,p'*-TDE, *o,p'*-TDE, *p,p'*-DDE) occurred from the resort area with a peak of 5.4 kg (11.8 lb) total DDT per week and an average of 0.86 kg (1.9 lb) total DDT per week. Corresponding figures from the urban-agricultural area were peak 1.1 kg (2.5 lb) and average 0.18 kg (0.40 lb) total DDT per week, and from the agricultural area peak 0.23 kg (0.50 lb) and average 0.09 kg (0.20 lb) per week. Differences can be explained partly by differences in techniques of application. In the agricultural area, DDT was incorporated into the soil, while in the resort area it was applied by ground or air application. Furthermore, the resort area is practically devoid of true soil and the pesticide accumulated mostly on surface detritus and moved easily with surface runoff.

Pesticide residues reaching aquatic environments in an adsorbed state can remain adsorbed with eroded soil particles and become part of the sediment. Residues in solution can be adsorbed by particulate matter including microbial flocs in the water. Thus, the sediment and particulate matter in water is a predominant feature controlling the accumulation and fate of pesticides in waterbodies (36). Microbial flocs are known to concentrate pesticides from water (106). Factors affecting sediment-pesticide-water interactions and pesticide persistence in aquatic systems were reviewed recently by Pionke and Chesters (144). Although adsorption enhances decontamination of the overlying water, it simultaneously increases the build-up of pesticides in the bottom sediment.

3.6 Natural Renovation Mechanisms Available to Remove Pesticide Residues in Soils and Aquatic Environments

Pesticides remaining in soils and those present in waterbodies are subjected continually to dissipation processes acting either singly or

interactively. The mechanisms of dissipation include adsorption, degradation (photo, chemical and microbial), plant uptake, volatilization, leaching, and surface runoff. The last three are means by which pesticides move from one segment (i.e., agricultural watersheds) of the ecosystem to another (atmospheric and aquatic systems), and these are discussed in Section 3.5. Dissipation processes except leaching and runoff are similar for both soil and aquatic environments. However, rates of pesticide dissipation may vary because of the differences in environmental conditions in the two systems. While pesticides have inherent but varying degrees of persistency because of their chemical nature, their longevity in these environments is modified by edaphic, climatic, and limnological factors (144). In this section, pesticide losses resulting from adsorption, degradation, and plant uptake are presented, followed by a summary of their persistence in soil and aquatic systems.

3.6.1 Adsorption

The effect of adsorption on pesticide movement is discussed at length in Sections 3.5.1, 3.5.2 and 3.5.3. Adsorption is an indirect mechanism of dissipation from the environment because it affects degradation rates and reduces pesticide bioactivity and volatility. Degradation rates of pesticides may be decreased or increased by adsorption depending upon the mechanism by which the compound is degraded. Adsorption by soil or sediment can stabilize pesticides against biodegradation while, in some cases, chemical degradation is accelerated. Furthermore, the availability of adsorbed pesticides to plants may be reduced markedly.

3.6.2 Photodecomposition

Photochemically-induced degradation occurs at surfaces, and pesticides residing at the soil surface are more subject to photodegradation than those incorporated into the soil. Additionally, photodecomposition has been observed in aqueous environments. Reviews on pesticide photodecomposition are available (40, 41, 42, 145). The practical significance of photodegradation

as a means of pesticide removal in soil and aquatic systems has not been determined quantitatively because of the difficulty of interpolating data gathered in the laboratory to field conditions. In natural systems, photochemical reactions occur simultaneously with adsorption and microbial decomposition, but in most cases the photodegradation reactions are comparatively slow. However, the possibility exists that photodecomposition contributes to the airborne dissipation of some pesticides.

Photochemical degradation has been demonstrated for many pesticides including a number of organochlorine insecticides. DDT was found to degrade slowly in sunlight to DDE, TDE (DDD) and other products (146). Aldrin and dieldrin, upon exposure to sunlight, were converted to photoaldrin and photodieldrin, respectively (153). Photoaldrin was approximately twice as toxic as the parent compound to insects and mice. Other related insecticides photolyzed by sunlight include chlordane, endrin, heptachlor, isodrin and methoxychlor.

3.6.3 Plant Uptake

The use of trap plants to remove pesticides from soil and water systems has been suggested (55). This potential environmental cleanup method was based on the fact that many terrestrial and aquatic plants are capable of absorbing and translocating pesticides (45, 96), followed by possible detoxification of the compound to less active components.

Several terrestrial crops were found to absorb chlorinated insecticides (17, 110, 113, 132, 135), and evidence of metabolic breakdown was indicated for DDT, heptachlor, endrin, γ-BHC and aldrin. Corn, which is resistant to the *s*-triazine herbicides--atrazine and simazine--absorbed these herbicides from soils and metabolized them to nonphytotoxic compounds (129). Dissipation of atrazine from soils through uptake by corn, sorghum and johnsongrass has also been studied (169).

Many species of algae and aquatic plants were able to remove low concentrations of herbicides, namely, 2,4-D, amitrole, atrazine, dicamba, dichlobenil and diphenamid, from water (21). Once absorbed, most of the

pesticides were metabolized by various aquatic plants and algae. Metabolism was rapid, particularly with the algae. In some instances, portions of the metabolites and undegraded herbicides were released back to the water. In another investigation it was shown that algae were able to concentrate pesticides by a factor of severalfold and were generally more resistant to pesticide toxicity than higher members of the food chain. This suggests that phytoplankton and aquatic weeds can remove effectively the low-level concentrations of pesticides normally encountered in aquatic systems and metabolize them to less active compounds.

Sorption of herbicides by weeds results frequently in the death of the plant. If degradation of the herbicide is slow, the sorbed herbicide may be returned to the soil or lake sediment where the dead weeds undergo decomposition. For example, in an aquatic weed control program, using paraquat and diquat, the herbicides were not detected in the sediment until the dead weeds had settled to the bottom sediment and had been subjected to decomposition processes (57). Upon decay of the dead weeds the herbicides were either released and adsorbed by the sediment or remained bound with the settled organic material. In addition, certain pesticides may be absorbed and translocated without degrading as shown in the case of dieldrin uptake by corn (19), or simply may be converted to another active compound as was found for enzymatic conversion of aldrin to dieldrin in peas (110). Plant-absorbed pesticides which are resistant to metabolic breakdown within the plant either become part of the food chain or are returned to the soil or sediment.

Probably the extent of pesticide detoxification by plants is small, as uptake is limited by spatial availability of the pesticides and by the sorption capacity of the plant. However, a great need exists to better understand the mechanism of pesticide dissipation through plant uptake. Included in these investigations must be a search for terrestrial and aquatic plants that are efficient in detoxifying a wide spectrum of pesticides absorbed by them.

3.6.4 Chemical Degradation

Nonbiological processes of pesticide breakdown in soils and sediments have long been recognized. Chemical reactions of pesticides may occur independently of soil or they may be soil-catalyzed. Extensive coverage of this topic is provided in the comprehensive review by Helling et al. (82).

The chemical hydrolysis of organochlorine insecticides has not received as much attention as that of the organophosphorous insecticides and herbicides. Evidence for the chemical decompositions of chlordane, toxaphene, heptachlor, DDT, dieldrin and endrin have been reported in dry, acidic clay diluents used in insecticidal formulations (53). Chemical conversion of endrin to two isomers--ketone and aldehyde--was observed in dry, acidic soils but no conversion occurred in the presence of moisture (53, 134). DDT conversion to DDE has been shown to occur in moist and dry soils ranging in texture from sandy loam to clay under a wide range of pH values (136). Increasing the soil pH by liming and additions of Fe_2O_3 and Al_2O_3 to the soil failed to enhance conversion of DDT to DDE. Data indicate that the predominant mechanism of conversion of DDT in moist soil is by a microbial pathway, while chemical degradation is predominant in dry soil. Similar findings have been reported for heptachlor (23). Conversion of heptachlor to 1-hydroxychlordene was rapid in a series of dry soils of low organic matter content. The reaction did not proceed in soils of high organic matter content.

Results of these investigations indicate that chemical breakdown may play a significant role in the dissipation of soil-adsorbed organochlorine insecticides during dry periods. However, much more information is required on the rates of these conversions and on the environmental conditions which promote this type of organochlorine insecticide dissipation from the environment.

Several organophosphorous insecticides and herbicides have been shown to degrade rapidly by chemical hydrolysis. The process is catalyzed by adsorption at soil colloid surfaces and follows first-order kinetics (7, 8, 98, 99, 100). In soil systems, chemical hydrolysis of malathion

was rapid and was almost completed before the termination of the lag phase necessary for promotion of the biological degradation mechanism for malathion (100). Alkaline conditions in the soil enhanced degradation of malathion by chemical hydrolysis (186). Other organophosphorous pesticides found to degrade chemically are diazinon (99, 164), phorate (61), dichlorvos (62), ciodrin (98), and imidan (122). The longer-lived organophosphorous pesticides, including methyl parathion, parathion (62, 63, 165), dimethoate, zinophos, and dursban (62) are degraded primarily by microbial mechanisms.

Chemical hydrolysis of the 2-chloro-*s*-triazine herbicides in soils and sediments has been reported (7, 8, 77, 171). The hydrolysis of atrazine to nonphytotoxic hydroxyatrazine is enhanced by atrazine adsorption possibly to carboxyl groups present on the organic components of soils and sediments (7).

Although certain pesticides are able to undergo rapid chemical degradation, formation of degradation products as persistent, or more so, as the parent compound may occur. For instance, among the degradation products of diazinon is diethyl thiophosphoric acid which is more stable in soils than the parent compound (100). A gap still exists in the understanding of the chemical mechanisms of degradation of the more persistent pesticides, particularly the organochlorine insecticides, as well as the fate and toxicity of the resulting degradation products.

3.6.5 Microbial Degradation

Microbial metabolism is considered to be the major pathway of degradation for many pesticides in soils and sediments. The efficiency of this pathway depends upon such environmental factors as temperature, moisture content, organic matter content, aeration, pH, and pesticide concentration. Although intensive studies have been made on the mechanisms by which microorganisms degrade pesticides, the processes are not understood clearly.

In general, the organochlorine insecticides are the most resistant pesticides to microbial attack. Degradation products encountered frequently

are compounds which retain insecticidal properties. For example, partial degradation of DDT results in the formation of TDE (DDD) and DDE. TDE (DDD) and DDE are stable in soil and aquatic systems, and their metabolic fate in these environments remains relatively unknown. Several organochlorine insecticides although relatively persistent may degrade at a potentially significant rate in soils. Heptachlor (126), lindane (209), and endrin (23) have been shown to degrade in soils to compounds of reduced insecticidal activity. Conversion of heptachlor to the less toxic metabolite 1-hydroxychlordene also has been reported (126). The oxygen status of soils and aquatic systems has pronounced effects on the microbial breakdown of many organochlorine insecticides. In soils, DDT is rapidly converted to TDE (DDD) under anaerobic conditions and very slowly to DDE under aerobic conditions (68, 69, 95). The addition of such readily available energy sources as alfalfa meal or sucrose promoted the anaerobic disappearance of DDT from soils (26, 69). However, in aerobic soils DDT remained stable despite the addition of alfalfa meal. These observations suggest that flooding DDT-contaminated soils may accelerate decontamination. Conditions conducive to anaerobiosis, like thermal stratification of lakes, may also play an important role in the disappearance of DDT accumulated in bottom sediments. In raw water containing some colloidal material and maintained under aerobic conditions, heptachlor and endosulfan were degraded rapidly while lindane, heptachlor epoxide, dieldrin, DDT, TDE (DDD), DDE and endrin were not degraded (47). Endrin, *p,p'*-DDT, lindane, aldrin, and heptachlor were degraded in anaerobic digested wastewater sludge to various derivatives (83). Although conditions in the sludge are not characteristic of most lakes, the data emphasize the role of oxygen deficiency in the metabolic fate of many persistent pesticides.

The conversion of *p,p'*-DDT to *p,p'*-DDD has been observed in flooded anaerobic soils (34, 69) and in oxygen-deficient lake water (128). The time required to convert 50% of the applied DDT to TDE (DDD) was 24 hr in the lake water and 8 weeks in the flooded soils. However, TDE (DDD) formed in flooded soils seemed to resist further degradation (34, 71). Anaerobic decomposition of DDT to TDE (DDD) has been observed in lake

sediments, but the degradation product TDE (DDD) appears to be relatively stable in lake sediments (120, 140). The rapid degradation of lindane has been reported in flooded rice soils (208), lake muds (144), and in simulated lake impoundments (137). It was estimated in the simulated lake impoundment that only 15% of the lindane was degraded in the aerobic system in 90 days while degradation was almost complete in the anaerobic system in the same time period (137). The major intermediate product volatilized rapidly. In addition, methoxychlor and heptachlor were also found to degrade in submerged soils, and the rate of loss was accelerated in the presence of high organic matter content (34).

As stated earlier, many organophosphorous insecticides are hydrolyzed rapidly by nonmicrobial means. However, for the more persistent types, microbial breakdown may predominate. Thus, conditions retarding microbial activity may result in enhanced persistence of the organophosphorous insecticides.

Soil and lake sediment microorganisms have been implicated in the degradation of several organophosphorous insecticides including diazinon (162), phorate (114), malathion (119, 186), and parathion (63, 111, 165). The degradation of diazinon (164) and parathion (165) was considerably increased in flooded soils. Apparently, the presence of molecular oxygen retarded microbial degradation although this was not observed for parathion in lake sediments (63). The degradation of parathion to aminoparathion was found to be greatly accelerated by microbial activity, and the rate was similar under aerobic and anaerobic conditions. These results imply that in sediments aerobic and anaerobic organisms are able to degrade parathion. Adsorption of some organophosphorous insecticides may increase their persistence in aquatic systems. For example, dursban was shown to persist for much longer periods of time in aquatic systems of high colloidal organic matter content due to pesticide adsorption (156).

Microbial degradation of organochlorine and organophosphorous insecticides in aquatic environments was reviewed recently (141, 163). In a review of the degradation of the newer vinyl phosphate insecticides, such as phosphamidon, chlorfenvinphos, and mevinphos, it has been shown that

the half-life of these pesticides in soils ranges from 1 week to 7 months (20). Chlorfenvinphos appeared to be the most resistant to biological decomposition.

Despite the increasing use of the carbamate insecticides, investigation of their degradation has not been as extensive as for other insecticidal groups (158). Carbaryl, the most widely used carbamate, was found to be readily degraded by microorganisms in soils (116) and natural waters (6). Some members of the group, such as dimetilan and pyrolan, tended to be more biologically stable in natural waters possibly due to their strong adsorption on suspended clay minerals (6).

Most of the agriculturally-used herbidides are degraded primarily by microorganisms. A great deal of study has been devoted to their degradation and exhaustive reviews have been compiled (92). Many of the herbicide groups, including the thiocarbamate (93, 172), phenylcarbamate (91), substituted urea (91, 94), and the commonly used chlorinated aliphatic acids (54) and phenoxyalkanoic acids (115), are microbiologically degraded in soils in relatively short periods. Compared to organochlorine insecticides, phenoxyalkanoic acids, such as 2,4,5-T and particularly 2,4-D, degrade rapidly in soils and sediment-water systems (55, 160). Anaerobic conditions tend to retard 2,4-D metabolism, indicating the importance of oxygen in the metabolism of this compound in the environment (104). Several of the benzoic and phenylacetic acid herbicides are more persistent. For instance, dicamba, 2,3,6-TBA, fenac and methoxyfenac are comparatively more resistant to microbial attack than chloramben (166) which is degradable readily by microbial processes (198). Picloram and trifluralin are considered to be persistent herbicides (39). In a long-term field experiment picloram residues were detectable 9 to 15 months after application (123). Propanil, an anilide herbicide, was found to be microbially-degraded primarily to chloranilines (13). However, the metabolites are adsorbed strongly by soil organic matter and may exist in the soil for several years. Some of the easily degradable herbicides may persist longer in aquatic than in terrestrial systems. The substituted urea herbicides, monuron and neburon, persisted in excess of 2 years in simulated ponds

containing amorphous materials (56) while CIPC--a carbamate herbicide--persisted under anaerobic conditions induced in an artificial impoundment (161).

The *s*-triazine herbicides, as indicated earlier, can be extensively degraded chemically in soils or sediments, but they are also subject to microbiological metabolism (91). Atrazine, the most widely used *s*-triazine, is quite stable compared with most of the herbicides and can persist in soil for more than one growing season (27, 104). Rates of atrazine degradation are influenced greatly by the spatial location of the herbicide in the soil profile. Atrazine present in the plow layer persisted for 5 months while atrazine at 40- and 90-cm (15.7- and 35.4-in) depths persisted for 17 months and 41 months, respectively (104), which suggests the importance of management techniques in minimizing atrazine leaching.

Adsorption of pesticides by clay minerals may protect them from microbial attack. The dipyridyl herbicides--paraquat and diquat--are shown to be metabolized by soil microorganisms (59), but once adsorbed by clay minerals they become extremely resistant to microbial decomposition (191). In aerobic and anaerobic aquatic systems, diquat was not degraded within 6 months (170).

Figure 4 summarizes the persistence of 11 major pesticide groups (95). The persistence values represent a 75 to 100% loss of the pesticide. As indicated elsewhere in this section, persistence of pesticides in aquatic environments may differ from that in terrestrial soils, but more investigations are needed to make generalizations. Furthermore, to elucidate the metabolic products, sustained and continuing research is needed in order to better understand the environmental implications of pesticides, particularly in the aquatic segment of the ecosystem.

3.7 Probable Changes in Land Use Activities with Time

Tables 1 and 2 show that a general decrease in agricultural land area in the U.S. portion of the Great Lakes Basin will occur for the next 50 years. However, the decline is projected to be slight, at approximately 5% for cultivated cropland and 8% for pasture land. If predictions by crop

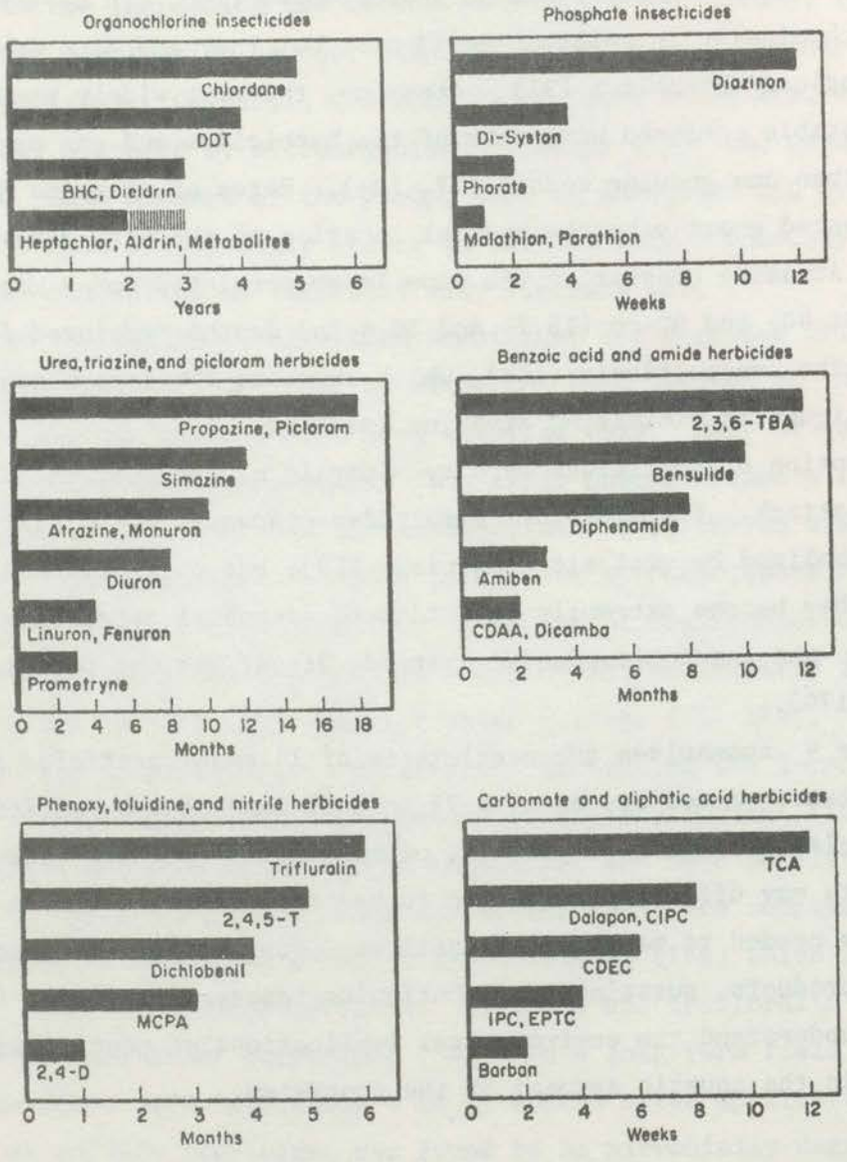

Fig. 4. Persistence of pesticides in soils. Redrawn from Kearney et al. (95).

category are considered, area for row and specialty crops will increase by 22 and 5%, respectively, as a result of projected increases in areas sown to corn, soybeans and commercial vegetables. As urbanization expands, thereby placing increased pressure on cultivated and pasture lands, some of the present idle cropland will likely be brought into cultivation. By the year 2020 it is estimated that the area of idle cropland will decrease by about 26%.

Estimated population increases in the region (65) will exert a marked influence on crop production. Since the projected area devoted to crop production is substantially unchanged, higher crop production per unit area will be needed to meet the food requirements of a rapidly-expanding population by the year 2020. Simultaneous growth of the livestock industry will probably follow the population trends, making it necessary also to increase feed production per unit area. The tremendous increase in the projected crop-yield increase per unit area is illustrated in Fig. 5. The figure suggests clearly that the area presently under cultivation will be cropped much more intensively to double or even triple the current level of production.

In order to sustain the high level of crop yields, the use of more pesticides is inevitable. Alternative methods of control for most pests are still not available. Although research on alternative methods are continuing, development of a widely-accepted and economically-feasible method may not be available in the near future. In the Great Lakes Region, pesticide usage will increase, particularly in the case of herbicides, as indicated previously in Section 3.1.2. The use of organochlorine insecticides is curtailed greatly due to the numerous regulatory restrictions placed on them. However, their use is being replaced by the use of organophosphorous and carbamate pesticides. The quantities of these compounds used for agricultural purposes are expected to increase, and an expanded research program on the stability, degradability and toxicology of the carbamate and organophosphorous insecticides is required. Furthermore, as new pesticides are introduced, a capability should be developed so that the impact of these compounds on the environment can be predicted.

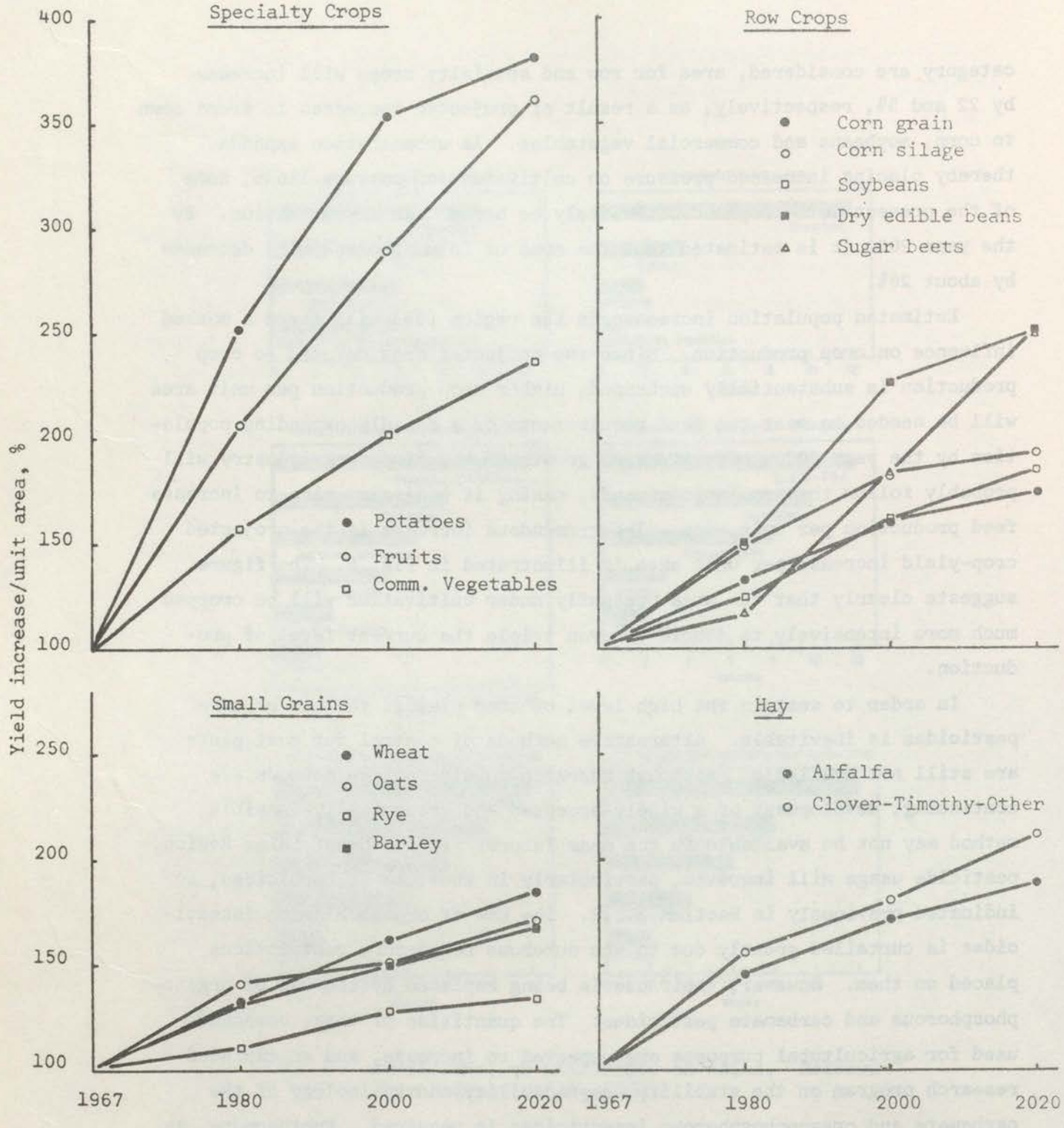

Fig. 5. Projected yield increases per unit area of major crops in the Great Lakes Region, U.S. for 1980, 2000, 2020 using 1967 as the base year. (Assembled from data in Table 1.)

3.8 Projected Seriousness of Future Pesticide Pollution Problems

3.8.1 General Level of Pesticide Contamination of Agricultural Soil and the Great Lakes

In order to assess the seriousness of pesticide pollution, it is important to know the extent of pesticide contamination of the soil and aquatic environments. However, only general indications of pesticide pollution can be made because adequate assessment is undoubtedly difficult to undertake.

Studies discussed in Section 3.5.3 indicate that the absolute amount of pesticide residues lost from agricultural watersheds through runoff is usually in the µg/l (ppb) range. Portions of the pesticides applied are associated with the soil, and the longevity of these residues is dependent upon the rates of natural renovation mechanisms and nature of the chemical. A summary of the residue content of the most commonly used organochlorine insecticides in several agricultural soils of the United States were summarized by Edwards (45). On the average, most agricultural soils contained between 1 to 4 mg/kg (1 to 4 ppm) DDT, including analogs, with the exception of orchard soils which contained substantially higher amounts, ranging from 19 to 74 mg/kg (19 to 74 ppm). Concentrations of γ-BHC, aldrin, dieldrin, chlordane, and heptachlor were generally less than 1 mg/kg (1 ppm). Pesticide residues in cropland soils for 43 states, reported by the National Soils Monitoring Program, are shown in Table 13 (197). Fruit and vegetable soils contained, on the average, the highest level of DDT of 1.92 mg/kg (1.92 ppm), which is six times higher than the national average of 0.31 mg/kg (0.31 ppm). The DDT-metabolites--DDE and TDE (DDD)--were also highest in these soils. These high levels are caused by repeated application of DDT in many types of orchards and confirms the results summarized earlier in this section. Other than orchard soils, residues found in cotton and vegetable soils were highest, while the lowest amounts were found in soils under corn and small grains. The corn, vegetable, and vegetable and fruit soils had the highest residues of dieldrin,

TABLE 13

Mean pesticide residues in mg/kg (ppm) in soil for various cropping regions in the United States, FY 1969. (Adapted from Wiersma et al. (197).)

Compound	Corn	Cotton	Cotton and General Farming	General Farming	Hay and General Farming	Irrigated Land	Small Grains	Vegetable	Vegetable and Fruit	Range of Detected Residues	No. of Samples Analyzed
Aldrin	0.05	<0.01	<0.01	0.02	<0.01	<0.01	<0.01	<0.01	0.01	0.01-3.06	1,729
Arsenic	7.44	6.72	4.88	5.35	6.42	4.77	5.70	8.75	3.27	0.25-107.45	1,726
Atrazine	0.02						<0.01			0.01-1.55	199
Carbophenothion			-	-		-				0.23	66
Chlordane	0.09	<0.01	0.01	0.01	0.03	0.03	<0.01	<0.01	0.14	0.01-6.30	1,729
2,4-D				<0.01			<0.01	-		0.01-0.03	188
DCPA	-	-	-	-	-	<0.01	-	-	-	0.54	1,729
o,p'-DDE	<0.01	0.01	<0.01	<0.01	<0.01	0.01	-	<0.01	0.01	0.01-0.20	1,729
p,p'-DDE	0.01	0.16	0.13	0.07	0.05	0.18	<0.01	0.18	0.37	0.01-6.99	1,729
o,p'-DDT	0.01	0.09	0.04	0.05	0.03	0.05	<0.01	0.07	0.06	0.01-6.29	1,729
p,p'-DDT	0.06	0.54	0.22	0.25	0.20	0.19	<0.01	0.50	0.64	0.01-35.92	1,729
DDTR	0.14	0.87	0.44	0.43	0.30	0.48	<0.01	0.81	1.92	0.01-78.36	1,729
DEF	-	-	-	<0.01	-	-	-	-	-	0.12	1,729
Diazinon		-	-			0.01				0.02-0.15	66
Dicofol	<0.01	-	-	-	<0.01	0.01	-	-	-	0.03-1.07	1,729
Dieldrin	0.05	0.01	<0.01	0.03	0.02	0.02	<0.01	0.05	0.04	0.01-1.60	1,729
Endosulfan (I)	<0.01	-	-	-	<0.01	<0.01	-	-	-	0.01-0.24	1,729
Endosulfan (II)	<0.01	-	-	-	<0.01	0.01	-	<0.01	-	0.01-0.53	1,729
Endosulfan sulfate	<0.01	-	-	-	<0.01	0.01	-	<0.01	-	0.01-0.94	1,729
Endrin	<0.01	<0.01	<0.01	<0.01	-	0.01	<0.01	0.01	0.01	0.01-0.56	1,729
Endrin aldehyde	-	-	-	-	-	-	-	-	<0.01	0.03	1,729
Endrin ketone	-	0.01	<0.01	-	-	<0.01	-	<0.01	<0.01	0.01-0.13	1,729
Ethion		-	-			<0.01				0.03	66
Ethyl parathion		-	-	<0.01	-	<0.01				0.01-3.01	66
Heptachlor	0.01	-	<0.01	<0.01	<0.01	<0.01	-	-	<0.01	0.01-0.97	1,729
Heptachlor expoxide	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01-1.08	1,729
Isodrin	<0.01	-	-	<0.01	-	-	-	-	-	0.01-0.03	1,729
Lindane	<0.01	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	-	0.01-0.35	1,729
Malathion		-	-			-				0.04-0.36	66
Methoxychlor	-	-	-	-	-	-	-	<0.01	-	0.28	1,729
PCNB	-	-	-	<0.01	-	-	-	-	-	0.69	1,729
o,p'-TDE	<0.01	<0.01	<0.01	0.01	<0.01	0.01	-	0.01	0.15	0.01-4.52	1,729
p,p'-TDE	0.05	0.07	0.04	0.04	0.01	0.04	<0.01	0.05	0.70	0.01-31.43	1,729
Toxaphene	<0.01	0.42	0.20	0.16	-	0.14	-	0.01	0.08	0.10-11.72	1,729
Trifluralin	<0.01	0.01	<0.01	<0.01	-	0.01	<0.01	<0.01	<0.01	0.01-0.25	1,729

NOTE: Blank = not analyzed; - = not detected

i.e., approximately twice as high as the national mean of 0.03 mg/kg (0.03 ppm). These data represent the residues left in the soil which may have built up due either to repeated application or high intensity of application. Thus, the soil serves as an effective reservoir of pesticide residues and metabolites which are available potentially for further transport. Although pesticide residue levels in cropland soils of some agricultural watersheds of the Great Lakes Region are well-documented and investigated, no comprehensive effort has been made to monitor pesticide residues on a continuing basis.

Residues reaching the Great Lakes are low, ranging from ng/l (ppt) to µg/l (ppb) levels. This is clearly demonstrated by the results of monitoring studies conducted on lake and tributary waters (58, 103, 109, 127, 204). Although the amounts of pesticide reaching the Great Lakes are low-level residues, contamination is continual as a result of the perpetual process of runoff and erosion occurring from adjacent treated agricultural watersheds. Furthermore, the impact of pesticides on aquatic organisms is of major concern. Fish, mussels, shrimp, invertebrates, plankton, aquatic plants, and other aquatic organims, through the process of biomagnification, concentrate the low-level residues in their tissues (18, 32, 45, 103, 127, 147, 148, 185, 204). Clearly, the hazards of pesticides in the aquatic environment lie in the biological concentration process at all trophic levels in the food chain. Effects of food chain contamination on fish and wildlife have already been well-documented and reviewed (35, 45, 103, 130, 207). Pesticide residue accumulation in the food chain may also result in hazardous exposure to humans.

Current information reveals that among the Great Lakes, Lake Michigan has the greatest pesticide problem (105, 148, 149). In view of this, most of the monitoring programs have been directed at that lake. Recently, an evaluation of DDT and dieldrin in Lake Michigan was made based on monitoring waters, biological indicators, fish and sediments for these compounds (103). Concentrations of DDT and dieldrin in lake and tributary waters, as well as in stream sediments, are presented in Tables 14, 15 and 16, respectively. The levels of DDT and dieldrin in the waters are normally

TABLE 14

Lake Michigan Open Water Pesticides--July 1969. (Adapted from The Lake Michigan Interstate Pesticide Committee of the Lake Michigan Enforcement Conference (103).)

Sample No.	Location (1/2 mile off shore)	Total DDT	Dieldrin	BHC	Estimated PCB
		ng/l			
3	Little Suamico River	<1	1.1	111	2.5
4	Pensaukee River	2.6	2.6	140	6.5
5	Oconto River	12.7	1.3	50.3	
6	Peshtigo River	<1	<1	33.0	2.8
7	Menominee River	1.1	1.5	16.5	2.5
9	Clark Lake Creek	3.5	1.5	13.7	9.0
11	Kangaroo Lake Creek	<1	1.7	8.0	Off Scale
14	Ahnapee River	15.3	1.0	23.3	50.8
15	Kewaunee River	<1	1.1	8.5	
16	East Twin River	8.8	2.2	20.0	19.5
21	Pine Creek	31.2	2.6	7.5	Off Scale
26	Pigeon River	<1	<1	11.2	2.5
30	Kinnickinnic River	5.5	2.2	8.0	9.5
31	Menomonee River	12.0	1.1	20.2	15.0
32	Oak Creek	10.8	3.3	22.8	27.2
33	Root Creek	5.9	4.5	110	5.5
34	Pike River	2.1	3.1	27.8	
35	Barnes Creek	5.0	1.9	9.0	
36	Calumet River at Calumet City	4.1	2.1	7.8	55.9
37	Burns Ditch	4.7	2.3	8.3	9.3
38	Trail Creek	3.5	2.1	1.5	6.5
39	Galien River	10.1	3.4	4.5	15.0
40	Drain at Sawyer	1.5	1.9	7.4	
42	Paw Paw River	11.1	3.0	34.3	28.3
43	Black River	1.2	1.8	1.8	2.5
48	Muskegon River at mouth into Muskegon Lake	1.4	1.6	12.3	2.5
49	White River	1.4	1.5	18.1	2.0
50	Pentwater River	5.0	2.7	5.8	12.5
52	Manistee River	<1	<1	5.9	2.5
53	Betsie River	1.3	1.2	6.2	
54	Platte River	2.5	1.4	9.9	5.0
60	Bear River, Petoskey	1.5	1.3	2.1	
62	Manistique River	11.1	1.6	6.5	
65	Escanaba River	<1	<1	16.1	2.0

NOTE: ng/l = ppt; 1 mile = 1.609 km.

TABLE 15

Lake Michigan River Water Pesticides--July 1969. (Adapted from The Lake Michigan Interstate Pesticide Committee of the Lake Michigan Enforcement Conference (103).)

Sample No.	Location	Total DDT	Dieldrin	BHC	Estimated PCB
		ng/l			
2	Big Suamico River	<10	<10	NC**	NC
3	Little Suamico River	<10		NC	NC
4	Pensaukee River	<10	<10	NC	NC
5	Oconto River	<10	<10	NC	NC
6	Peshtigo River	<10	<10	73.5	NC
7	Menominee River	<10	<10	128	NC
8	Mud Lake Creek	<10		16.0	NC
9	Clark Lake Creek	600*	10	17.8	NC
10	N Jacksonport Creek	<10		43.3	NC
11	Kangaroo Lake Creek	14.4*	<10	52.2	NC
12	Moonlight Bay	<10	<10	27.2	NC
13	Stony Creek	<10	<10	35.3	NC
14	Ahnapee River	360*	<10	72.2	NC
15	Kewaunee River	<10	<10	13.3	NC
16	East Twin River	<10	<10	50.0	NC
17	West Twin River	27.6	<10	14.4	
18	Manitowoc River	<10	<10	66.7	NC
19	Silver Creek	10.8	<10	34.4	NC
20	Calvin Creek	<10	<10	<10	NC
21	Pine Creek	574*	<10	<10	NC
22	Point Creek	<10	<10	<10	NC
23	Fisher Creek	<10	<10	<10	NC
24	Centerville Creek	<10	<10	174	NC
25	Seven Mile Creek	542*	<10	20.8	NC
26	Pigeon River	<10	<10	<10	NC
27	Sheboygan River	260*	<10	5.2	Very possible
28	Black River	97.4*	<10	18.4	Very possible
29	Sauk Creek	29.8*	<10	19.5	Possible
30	Milwaukee River	42.8*	28.8	44	Possible
31	Menomonee River	53.8*	14	84	Possible
33	Root Creek	10.0	<10	64	Possible
34	Pike River	49.1*	<10	10.4	Possible
35	Barnes Creek	75.5*	<10	<10	Possible
36	Calumet River at Calumet City	50*	<10	40.4	
37	Burns Ditch	<10	<10	17.9	NC
38	Trail Creek	93.6*	<10	47	Possible
39	Galien River	56.9*	<10	14	Possible
40	Drain at Sawyer	16.0	<10	40	
41	St. Joseph River	<10	<10	23.0	30.0
42	Paw Paw River	36.0*	45.2	7.2	
43	Black River	82.2*	<10	<10	NC
44	Kalamazoo River	<10	<10	33.0	NC
45	Black River	<10	<10	<10	<10
46	Pigeon River	<10	<10	10.0	NC
47	Grand River	11.4*	<10	<10	Possible
48	Muskegon River at mouth into Muskegon L.	<10	<10	<10	NC
49	White River	<10	<10	<10	NC
50	Pentwater River	<10	<10	<10	<10
51	Pere Marquette River	<10	<10	20	NC
52	Manistee River	24.4*	<10	<10	Possible
53	Betsie River	<10	<10	<10	NC
54	Platte River	<10	<10	<10	NC
56	Leelanau Lake	<10	<10	<10	NC
57	Boardman River	<10	<10	Inter	NC
59	Lake Charlevois Outlet	600*	<10	<10	NC
60	Bear River, Petoskey	<10	<10	NC	NC
61	Millecoquins Creek	<10	<10	NC	NC
62	Manistique River	<10	<10	<10	NC
63	Sturgeon River	<10	<10	<10	NC
64	Whitefish River	975*	<10	NC	NC
65	Escanaba River	<10	<10	<10	NC
66	Ford River	<10	<10	<10	NC

*Retention time not exact
**Not calculated
NOTE: ng/l = ppt

TABLE 16

Pesticides in Stream Sediments-July 1969. (Adapted from The Lake Michigan Interstate Pesticide Committee of the Lake Michigan Enforcement Conference, 1972 (103).)

Sample No.	Location	Total DDT	Dieldrin	Estimated PCB
		µg/g*		
1	East River	1.07	0.001	0.50
2	Big Suamico River	0.001	0.001	0.01
3	Little Suamico River	0.006	0.001	0.02
4	Pensaukee River	0.023	0.001	0.05
5	Oconto River	0.002	0.001	0.01
6	Peshtigo River	0.002	0.001	0.01
7	Menominee River	0.001	0.001	0.01
8	Mud Lake Creek	0.001	0.001	0.01
9	Clark Lake Creek	0.011		0.028
10	N Jacksonport Creek	0.003	0.001	0.01
11	Kangaroo Lake Creek	0.015	0.001	0.01
13	Stony Creek	0.019	0.001	0.03
14	Ahnapee River	0.102	0.001	1.10
15	Kewaunee River	0.033	0.001	0.035
16	East Twin River	0.079	0.001	0.21
17	West Twin River	0.035	0.001	0.052
18	Manitowoc River	0.037	0.001	0.12
19	Silver Creek	0.016		0.01
20	Calvin Creek	0.082	0.001	0.01
21	Pine Creek	0.008		0.033
22	Point Creek	0.026	0.001	0.015
23	Fisher Creek	0.042	0.001	0.021
25	Seven Mile Creek	0.087	0.001	
26	Pigeon River	0.053		0.036
27	Sheboygan River	0.173	0.001	7.2
28	Black River	0.011		0.052
29	Sauk Creek	0.067	0.001	0.065
30	Milwaukee River	0.082	0.040	3.2
31	Menomonee River	0.114	0.040	4.4
33	Root Creek	0.069	0.004	0.075
34	Pike River	0.137	0.002	0.20
35	Barnes Creek	0.010		0.013
36	Calumet River at Calumet City	0.063	0.004	1.25
37	Burns Ditch	0.017	0.001	0.021
38	Trail Creek	0.143	0.002	
39	Galien River	0.024	0.002	0.060
40	Drain at Sawyer	0.009		0.037
41	St. Joseph River	0.029	0.001	0.032
42	Paw Paw River	0.035	0.001	0.08
43	Black River	0.044	0.005	0.11
44	Kalamazoo River	0.053	0.001	0.043
46	Pigeon River	0.03	0.001	0.018
47	Grand River	0.061	0.002	0.17
48	Muskegon River at mouth into Muskegon Lake	0.006	0.001	0.01
49	White River	0.008	0.003	0.02
50	Pentwater River	0.001	0.001	0.01
51	Pere Marquette River	0.007	0.001	0.01
52	Manistee River	0.001	0.001	0.01
53	Betsie River	0.012	0.001	0.01
54	Platte River	0.011	0 001	0.01
56	Leelanau Lake	0.005	0.001	0.01
57	Boardman River	0.008	0.001	0.01
59	Lake Charlevoix	0.008	0.001	0.01
60	Bear River, Petoskey	0.006	0.001	0.01
61	Millecoquins Creek	0.002	0.001	0.01
62	Manistique River	0.114	0.001	0.80
63	Sturgeon River	0.003	0.001	0.01
64	Whitefish River	0.007		0.04
65	Escanaba River	0.069	0.001	1.46
66	Ford River	0.007	0.001	0.03

*Wet basis

NOTE: µg/g = ppm

low, most samples containing less than 10 ng/l (10 ppt). However, the stream sediments contained several hundred times more DDT and dieldrin than the associated tributary waters. Nonetheless, pesticide monitoring in lake sediments is often neglected although sediments serve as the most important repository of pesticides in aquatic ecosystems (105). Eroded soil particles containing adsorbed pesticides eventually settle on lake bottoms and become part of the sediment. In addition, pesticides in water or in aquatic organisms may return to the sediment through sorption and decomposition. These situations underscore the importance of including stream and near-shore sediment sampling in any monitoring program of pesticides. The pesticide pollution of Lake Michigan, as indicated by high accumulations of DDT and dieldrin in sediment, fish, and biological samples, suggests higher usage of pesticides in the watershed of this lake than of the other lakes. One of the suspected sources is from treated agricultural watersheds; however, definitive studies of the contribution of farming activities on the Great Lakes Basin are yet to be undertaken.

3.8.2 Projections and Assumptions

The pesticide contamination of Lake Michigan has already been well-established. Pesticide input to the Great Lakes will continue as long as pesticides are used in the watersheds. As indicated in Section 3.7, pesticide use is predicted to increase due to projected increased crop and livestock production per unit area and the absence of suitable alternatives to replace chemical pest control. Even with curtailment in the use of DDT and other organochlorine insecticides, residues of these chemicals still present a continuing hazard to the aquatic environment due to their long persistence in soils. Evidence indicates that there is already a "pool" of DDT in soils, especially in heavily treated areas such as orchards. It is estimated that, if the use of DDT was to cease in the mid-1970's, detectable levels will still be found in the troposphere and the oceanic mixed layer in the year 2000 (207). Build-up of residues from herbicides may result due to increasing use and repeated applications. Commonly used herbicides like the s-triazines are known to persist in soils for a year or more. The less

persistent organophosphorous and carbamate insecticides may pose less of a hazard from residues. However, because of their rapid degradability in soils, more frequent applications are needed to obtain effective insect control. Metabolite accumulation in soil is relatively unknown except for DDE and TDE (DDD).

The seriousness of future pesticide pollution problems in the Great Lakes depends heavily on the adequacy of erosion and sediment control measures. Surface runoff--implicated as the main avenue of pesticide entry to aquatic systems--can be controlled effectively by present technology. However, putting the technology to work is often hampered by inadequate control measures and lack of effective implementation.

In the Great Lakes Region about 7,756,300 ha (19,165,900 acres) or 60% of the total area is currently in need of land treatment (Table 17). Using current conservation practices embodied in Public Law (P.L.) 566 and P.L. 46, only 40% is projected to be treated by the year 2020, leaving 60% untreated. Even accelerated land treatment schemes which involve cost-sharing between the Federal government and the farmer are projected to treat only 44% of the land by the year 2020.

It is interesting to note that in the above control legislation, the erosion and sediment problem has to be solved primarily on a voluntary basis without governmental units providing adequate regulations. These control programs are extremely slow and, if they continue to be implemented in this voluntary fashion, abatement of pesticide pollution of aquatic ecosystems will be very limited in the next 50 years. Hazards of erosion will be intensified by the opening of idle cropland since most of these lands lie on steep slopes or have shallow soils.

In order to minimize pesticide pollution effectively, mandatory sediment control programs in the Great Lakes Region are required urgently.

3.9 Review of Institutional Arrangements to Regulate Pesticide Pollution

The Federal Water Pollution Control Act Amendments of 1972 broadly regulate water pollution from all sources, including such non-point sources

TABLE 17

Total area, area treated and area needing treatment for 1970 and projected area that will be treated under current agricultural conservation practices and recommended accelerated programs in the U.S. Great Lakes Region, for 1980, 2000 and 2020. (Excerpted and recalculated from Appendix 13, Land Use and Management Work Group of the Great Lakes Basin Commission, sponsored by the Forest Service and Soil Conservation Service of the U.S. Department of Agriculture (66).)

Type of agricultural land	Total area 1970		Area treated prior to 1970		Area needing treatment after 1970	
	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3
Cropland	11,577.9	28,609.0	4,800.8	11,862.7	6,777.1	16,746.3
Pasture	1,418.8	3,505.8	439.6	1,086.2	979.2	2,419.6
TOTAL	12,996.7	32,114.8	5,240.4	12,948.9	7,756.3	19,165.9

	Projected area that will be treated under current conservation laws*						Area needing treatment after 2020	
	1980		2000		2020			
	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3
Cropland	650.1	1,606.4	1,950.3	4,819.3	2,730.7	6,747.8	4,046.3	9,998.5
Pasture	94.9	234.6	284.9	704.1	398.9	985.9	580.2	1,433.7
TOTAL	745.0	1,841.0	2,235.2	5,523.4	3,129.6	7,733.7	4,626.5	11,432.2
% **	9.6		28.8		40.4		59.6	

	Projected area that will be treated if the recommended accelerated programs are approved +						Area needing treatment after 2020	
	1980		2000		2020			
	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3
Cropland	779.9	1,927.2	2,145.0	5,300.4	2,990.6	7,389.9	3,786.5	9,356.4
Pasture	114.0	281.7	313.5	774.6	437.9	1,082.0	541.3	1,337.6
TOTAL	893.9	2,208.9	2,458.5	6,075.0	3,428.5	8,471.9	4,327.8	10,694.0
% **	11.5		31.7		44.2		55.8	

*Public Law 566 and P.L.-46 programs

**1970 used as base

+Programs providing funds for installation costs of land treatment measures

as agricultural land. In terms of pesticide pollution, specific federal and state laws and regulations have been enacted to control the registration, distribution, use and application of pesticides.

The Federal Environmental Pesticide Control Act (FEPCA) was enacted on October 21, 1972, amending the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) of 1947. Among the major provisions are included: registration, classification, applicator certification, intensified research and monitoring programs, development of disposal policies, and strengthening of enforcement policies. Most of the provisions of the act became effective immediately, while others have deadlines for later enforcement pending the establishment of regulations and development of federal standards to guide states in implementing the legislation. All provisions must be implemented by October 1976. Among the provisions to take effect by 1976 are: 1. classification of pesticides into general and restricted use, and 2. development, by states, of certification and licensing programs for applicators of restricted pesticides and subsequent approval of such programs by the U.S. Environmental Protection Agency (U.S. EPA). The act emphasizes the immediate need to develop biologically-integrated alternatives for pest control.

By authority of the FIFRA, the U.S. EPA Administrator after extensive hearings cancelled the nation-wide sale and use of DDT in June 1972 except for health reasons; the order became effective December 31, 1972. Use of other hazardous pesticides, such as 2,4,5-T, aldrin and dieldrin, have been cancelled temporarily, pending the results of public hearings.

The Great Lakes States have current laws or regulations which either conform with or are more stringent than the federal act. Given below are the laws and regulations by state. Most of the information given is based on that compiled by the Great Lakes Water Quality Board (67) and U.S. EPA (48).

3.9.1 Illinois

The Pesticide Control Law (Ch. 5, Ill. Ann. Stat. 1969) was passed

on June 25, 1969. This act provides for labeling of pesticides to prevent contamination of waters and the environment by regulating, restricting or prohibiting the sale, use or application of pesticides. It authorizes the Illinois Department of Agriculture to register and label all pesticides sold in the state. The agencies responsible for rules and regulatory programs are the Departments of Agriculture and Public Health.

Effective January 1, 1970, the Director of the Department of Public Health prohibited the sale, use or application of DDT without permit.

Illinois was among the first states to require the licensing of custom applicators of pesticides as provided in the Customs Application of Pesticides Act (Ch. 5, Ill. Ann. Stat. 1966).

3.9.2 Indiana

In 1971, Indiana enacted a law regulating the distribution, sale and use of pesticides and providing for the creation of a Pesticide Review Board (Sec. 15-2701, Ind. Ann. Stat. Supp. 1972). The law provides that all pesticides sold and transported in the state be registered with the Office of the State Chemist.

The Pesticide Review Board, which is composed of members from state agencies as well as from public and industry sectors, after public hearings, can regulate and adopt a list of restricted-use pesticides and pesticides for use by prescription only. Regulatory functions rest with the State Chemist and Pesticide Review Board.

The Indiana Stream Pollution Control Board works closely with the State Chemist and Pesticide Review Board to insure proper pesticide application in order to control pollution of streams in the Great Lakes Basin.

3.9.3 Michigan

Michigan promulgated the Economic Poison Law in 1967 (Sec. 12.352, Mich. Ann. Stat. 1967). This law specifies the registration and labeling of all economic poisons sold in the state. It further regulates the transportation, distribution, sale, and labeling of economic poisons. In the

same year, the Pesticide Applications Law (Sec. 12.353, Mich. Ann. Stat. 1967) was passed making it unlawful to apply pesticides commercially unless licensed by the Michigan Department of Agriculture.

An Economic Poisons Advisory Committee was formed in 1970. Under the interagency Agreement for Economic Poisons Label Review, each state agency makes periodic reviews of all labels of economic poisons proposed for registration.

In March 1972, a restricted-use pesticides regulation was promulgated for the identification of restricted-use pesticides. The restricted pesticides are to be sold only by licensed dealers to licensed applicators or representatives of governmental agencies. The Michigan Department of Agriculture assumes the regulatory responsibility for proper pesticide usage.

3.9.4 Minnesota

In 1969, the Economic Poisons and Devices Law (Sec. 24.069, Minn. Stat. 1969) was promulgated, creating the Pesticide Advisory Board which developed regulations on restricted-use pesticides issued by the Minnesota Department of Agriculture in mid-1970. These regulations restricted the sale, use and distribution of DDT and several other compounds. The agency responsible for registration and labeling is the Department of Agriculture while the Departments of Agriculture and Natural Resources assume the regulatory functions. Licensing of commercial applicators and dealers of restricted-use pesticides by the Department of Agriculture is provided in the Spraying and Dusting Law (Sec. 18.031, Minn. Stat. 1969).

3.9.5 New York

Enactment of the Economic Poisons Law in 1973 (Sec. 33-0101, N.Y. Environ. Con. Law, 1973) required that all pesticides used in the state be registered with the New York Department of Environmental Conservation. The department also has restricted the use of certain pesticides and prohibited the use of others. The Custom Application of Pesticides Act of 1973

(Sec. 33-0901, N.Y. Environ. Con. Law 1973) makes it unlawful to engage in custom application and sale of restricted-use pesticides unless registered and licensed by the Department of Environmental Conservation.

3.9.6 Ohio

Regulatory control of pesticide usage and application procedures is maintained by the Ohio Department of Agriculture. Pesticides must be registered with the department as provided by the state's Economic Poison Act (Sec. 921.11, Ohio Rev. Code, 1954) and Herbicide Law (Sec. 921.06, Ohio Rev. Code Supp. 1954). In addition, the Ohio Pesticide Use and Applicator Law (Sec. 921.41, Ohio Rev. Code Supp. 1971) requires the licensing of pesticide applicator firms as well as certification of pesticide equipment operators by the Department of Agriculture.

3.9.7 Pennsylvania

Registration of pesticides used in the state is embodied in the Pesticide Act of 1957.

3.9.8 Wisconsin

The basic Wisconsin pesticide use and control legislation was embodied in Chapter 94 of the Statutes. By authority of this law the Wisconsin Department of Agriculture has promulgated rules under Chapter Ag 29 of the Administrative Code to regulate registration and labeling of pesticides.

The Wisconsin Statutes of 1969 contained two laws on pesticides in Chapters 146 and 426. Chapter 426 prohibits the sale, use and transportation of DDT except under emergency conditions, making Wisconsin the first state to ban DDT by statute. Chapter 146 created the Pesticide Review Board, bringing three state departments--Agriculture, Natural Resources, and Health and Social Services--in close coordination in developing regulatory programs concerning pesticide use restrictions under the advisement of a Technical Advisory Council. The Administrative Code Chapters Ag 29 and NR 80 were

products of such interagency cooperation. Ag 29 lists certain prohibited-use pesticides and six restricted-use pesticides for agricultural and other purposes. NR80, promulgated by the Department of Natural Resources, lists restricted-use pesticides that may be used by permit only for wild animals.

Enforcement of statutes and rules is performed cooperatively between the Departments of Agriculture and Natural Resources although the latter has general jurisdiction.

3.9.9 Great Lakes Basin

The Great Lakes States appear to have adequate existing laws and regulations to meet the standards of the Federal Act concerning the sale, registration, distribution, use, and application of pesticides. Except for Wisconsin and Pennsylvania, most of the states already have passed the necessary enabling legislation requiring the licensing of commercial applicators of restricted-use pesticides. A proposed bill of this nature is pending in the Wisconsin Legislature.

The primary limitation in the success of pesticide regulations is in the monitoring and enforcement areas. Progress in any monitoring and enforcement program is curtailed severely by lack of funds and personnel. In addition, rapport between agencies responsible for enforcement programs is not always possible; thus, one single agency should be given sole authority for implementing all pesticide regulations.

One critical weakness of pesticide regulatory programs has been the inability to estimate quantities of pesticides used and location of their use. In order to have a sound pesticide management and control program, the state agencies concerned must establish a system of documenting the amounts and levels of pesticides used as well as information relating to when and where they were used. Furthermore, state agencies should initiate a vigorous effort to include soil and sediment in any monitoring program undertaken.

Successful curtailment of pesticide pollution from agricultural land can be achieved only if mandatory erosion- and sediment-control programs are enacted by all levels of state governments.

Section 4

SUMMARY OF RELEVANT ONGOING OR PROPOSED RESEARCH, DEMONSTRATION OR MONITORING PROGRAMS

4.1 Ongoing Research

There is a wealth of ongoing research covering many aspects of pesticides in relation to water quality. Basic research is pursued vigorously to gain better insight into the movement, degradation, and metabolic pathways of pesticides in soils as well as the fate and behavior of pesticides in aquatic systems. Analytical procedures are constantly undergoing modifications and improvements in order to better identify and quantify the pesticide residues and their metabolites in the environment. No attempt is made to list all the projects in progress, but a cross-section of the diverse research activities with particular reference to the Great Lakes Basin is presented in Table 18.

The assessment and quantification of the contribution of agricultural watersheds by runoff to the pesticide pollution of the Great Lakes are important aspects of the projects in progress (Table 18). In some projects, movement of pesticides from soils to aquatic systems will be evaluated by systems analyses in an attempt to develop a mathematical model for pesticide transport and runoff from agricultural watersheds. Monitoring and surveillance programs will continue to provide pesticide levels and rates of change of pesticide levels in tributary and lake waters.

The USDA is continuing support of projects related to alternative methods of pest control with the ultimate objective of minimizing residues in the environment. Alternative methods being pursued include biological control, plant resistance, and integrated approaches. Additionally, better pesticide management techniques, such as judicious methods of application and use of safer pesticides, are being investigated.

4.2 Proposed Projects

Two proposed projects which may have significant impact on Great Lakes water quality management follow (Table 19). These projects, including the

TABLE 18

Recent and current research on pesticides in the United States with particular reference to the Great Lakes Basin

Program Title	Investigator/Affiliation	Duration	Sponsor
Chemical and sediment movement from agricultural land into Lake Erie	G. O. Schwab Ohio State University		USDI
Evaluation of pesticide sources and levels tributary to Lakes Michigan and Superior	L. A. Lueschow Wisconsin Dept. of Natural Resources		U.S.EPA
Pesticide inputs and levels in Minnesota waters of the Lake Superior Basin	T. A. Olson University of Minnesota		U.S.EPA
Pesticide movement from cropland into Lake Erie	A. C. Waldron Ohio State University	1971-72	U.S.EPA
Pesticide runoff in the Great Lakes Basin	B. G. Ellis Michigan State University	1972-73	U.S.EPA
Quantification of pollutants in agricultural runoff	J. N. Dornbush South Dakota State University	1972-73	U.S.EPA
Development of pesticide transport and runoff model	N. Crawford Hydrocomp, Inc.	1972-73	U.S.EPA
Attenuation and runoff of pesticides from agricultural lands to surface waters	J. Box USDA-SWCRD Watkinsville, Georgia	1972-73	U.S.EPA
The study and use of soil parameters for describing pesticide movement through soils	J. M. Davidson Oklahoma State University	1972-74	U.S.EPA
Pesticide monitoring program - Lake Michigan and tributaries - Illinois	B. J. Leland Illinois Dept. of Public Instruction		U.S.EPA
Great Lakes pesticide monitoring program, Indiana	R. L. Carter Indiana State Board of Health		U.S.EPA
Monitoring of pesticide levels in the Great Lakes	R. Reinert U.S. Dept. of Commerce Lansing, Michigan		U.S. Dept. of Commerce
Pesticide monitoring of aquatic environment, Michigan portion of the Great Lakes Basin	C. H. Fetterolf Michigan Dept. of Natural Resources		U.S.EPA
Environmental implication of pesticide usage	Five universities cooperating on a regional project	1965-75	USDA-CSRS
Herbicide movement from application sites and effects on non-target species	Nine universities and agencies cooperating on a regional project	1970-76	USDA-CSRS
Pesticide mobility and degradation in soil-water systems	Nine universities cooperating on regional project	1965-74	USDA-CSRS
Analytical methods for pesticides in soil and water	F. A. Gunther University of California-Riverside		State of California
Development of analytical chemical methods for environmental and water quality	H. A. Salman U.S. Dept. of the Interior Denver, Colorado		USDI
Economic and social impact of adjustment in use of pesticides	R. B. Norgaard University of California-Berkeley	1971-76	USDA-CSRS
Weed control practices to reduce pollution	M. A. Ross Purdue University	1970-75	USDA-CSRS
Chlorinated pesticides in the soil-water plant systems and their management to avoid pollution	F. G. Viets, Jr. USDA-SWCRD Fort Collins, Colorado	1968-73	USDA-CSRS
Pesticide detoxication mechanisms of microorganisms in soil and water	J. M. Bollag Pennsylvania State University		USDA-CSRS
Ecology of pesticides in an aquatic ecosystem	R. C. Ball Michigan State University	1964-69	USDA-CSRS
Distribution of organophosphorus pesticide residues in natural waters and sediments	M. L. Yates U.S. Dept. of the Interior		USDI
Black Creek study, Maumee River Basin, Allen County, Indiana	Allen County, Soil and Water Conservation District	1973-77	U.S.EPA

TABLE 19

Proposed projects with reference to Great Lakes pollution

Program Title	Investigator/Affiliation	Duration	Sponsor
Watersheds Study - Task C International Joint Commission, Land Use Activities Reference Group	Several institutions and agencies of the states of Indiana, Michigan, New York, Ohio and Wisconsin and of the Canadian Federal Government and Ontario Provincial Government	1974-78	International Joint Commission through U.S. EPA and the Governments of Canada and Ontario
Development and Implementation of a Sediment Control Ordinance: Institutional Arrangements Necessary for Implementation of Control Methodology on Urban and Rural Lands	T. C. Daniel University of Wisconsin-Extension	1974-78	U.S. EPA

ongoing research in Allen County, Indiana, recognize the paramount importance of a multidisciplinary and multiagency approach to achieving solutions to water pollution problems arising from non-point sources. The major objectives of the IJC-Land Use Activities Reference Group watersheds study are to: 1. investigate the effects of land drainage on the pollutional input to the Great Lakes, 2. develop a predictive capacity with respect to the sources, forms and amounts of pollutants reaching the Great Lakes Basin, and 3. develop remedial measures for maintaining and improving water quality in the Great Lakes. The overall objectives of the project coordinated by the Wisconsin Board of Soil and Water Conservation Districts and the University of Wisconsin are to: 1. demonstrate the effectiveness of land control measures in improving water quality, and 2. devise the necessary institutional arrangements for the preparation, acceptance, adoption and implementation of a sediment control ordinance applicable to incorporated and unincorporated areas on a county-wide basis. Outputs from these projects, in addition to answering the objectives, will aid the participating U.S. federal and state and Canadian federal and provincial governments to implement portions of their water quality laws.

Section 5

NATURE AND AVAILABILITY OF TECHNOLOGY TO COPE WITH POLLUTION PROBLEMS

Technological competence is available to minimize pesticide contamination of soil and water, reduce pesticide transport to aquatic environments, and improve detection of pesticide pollution. However, success of a particular technology may be limited by the lack of mandatory regulations to implement it, such as is the case with erosion and runoff control programs.

It is important to minimize soil pesticide residues since the magnitude of pesticide pollution of aquatic ecosystems is related directly to the amounts present in soils. This is best done by reducing dosage through proper application methods. Drift loss does not only increase the amount of compound applied but also contaminates nontarget areas adjacent to application sites. Aerial spraying during favorable weather conditions should become a rule of thumb for applicators. Other means of reducing build-up of residues in the soil involve the employment of alternative methods of pest control and less persistent compounds. Practical alternative methods, however, are yet to be developed.

Most residues remain in the plow-layer depth either adsorbed or absorbed by particulate matter and are susceptible to transportation during occurrence of runoff. Accelerated erosion of the pesticide-soil complex can be minimized by judicious use of soil management and conservation practices. Federal and state agencies are involved in controlling erosion, but programs have been carried out on a voluntary incentive basis. While soil and water losses can be estimated using the universal soil loss equation, mathematical models for pesticide transport from agricultural watersheds need development. Such models would entail understanding the mechanisms of the pesticide-soil-biota-water interactions in the watersheds.

Monitoring of pesticide pollution in the Great Lakes is conducted by the U.S. federal agencies in cooperation with the lake states. In addition, the International Joint Commission and Canadian government conduct extensive

water quality assessment. Monitoring activities center mostly on the aquatic environment, but little or no research activity is being centered on residue detection in agricultural soils and in the atmosphere overlying the Great Lakes. Apparently, development of a technique to measure the amount of pesticide in the atmosphere should be one important aspect of the monitoring programs being undertaken.

Section 6

NEED FOR NEW RESEARCH, DEMONSTRATION OR MONITORING PROGRAMS

Pesticide pollution of the Great Lakes is well-established. Runoff from agricultural lands in the watersheds has been implicated as a major source of residues and/or metabolites. This suspicion is based largely on the fact that the major portion of pesticide usage occurs in crop production. It becomes imperative, therefore, to assess the contribution of agricultural lands to the total pesticide load of the Great Lakes and associated tributary rivers. Only very recently have there been attempts made to identify clearly the sources of pollutant input (including pesticides) to the Great Lakes through integrated multidisciplinary and multiagency approaches using large watersheds, as indicated in Section 4. These research and/or demonstration programs will take at least 5 to 10 years to be realized or initiated depending upon funding potential.

Specific research needs are as follows but not arranged necessarily in any order of priority.

a. Pesticide inputs on agricultural lands. This involves a systematic documentation of the lands and amounts of pesticides used, category of crops treated, and area treated for each crop category.

b. Pesticide degradation rates and fate of toxic metabolites. Little information is available on the fate of the metabolites of persistent pesticides in soil and aquatic systems. Likewise, information about the degradation of newer compounds, which are finding increasing use due to the restrictions placed on the persistent pesticides, is needed for the environmental conditions found in the Great Lakes Basin. Furthermore, the fate and toxicity of their metabolites need elucidation.

c. Transport of pesticides from agricultural watersheds to the Great Lakes. More information is needed on the mechanisms of pesticide movement from varied types of soils and cropping pattterns present in the watersheds. Inputs of pesticides to the Great Lakes by atmospheric fallout or rainout are often neglected.

d. Expansion and redesigning of monitoring programs. Sediments act as reservoirs of pollutants (including pesticides), but they are often overlooked in monitoring programs. Inclusion of soil, sediment, and atmospheric samplings in the monitoring programs is highly desirable. Critical evaluation of sampling, sample preservation and analytical techniques should be made as often as possible.

e. Search for improved pest control techniques. Since pesticide usage in crop production is inevitable in the foreseeable future, considerable research effort should be directed towards the development of pesticide management techniques, including application methods and development of safer pesticides compatible with efficient soil and crop management systems. However, continuing and new research techniques of pest control are needed to provide alternatives to chemical pest control.

Section 7

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Category A6 - Nutrients

POLLUTION OF THE GREAT LAKES BY NUTRIENTS FROM AGRICULTURAL LAND

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To be used as portion of the U. S. Task A Report on GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by the International Joint Commission - prepared in partial fulfillment of U. S. Environmental Protection Agency Contract No. 68-01-1598

July 1974

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Section 1

SUMMARY

1.1 Introduction

The land use category on Nutrients - Agricultural Land is concerned with the nutrient contribution to the Great Lakes from agricultural land. The nutrients of importance are phosphorus and nitrogen, as algal growth is considered to be limited by one or the other of these nutrients in most lakes. Phosphorus reduction has been emphasized in municipal waste treatment. Consequently, assessment of the phosphorus loading from agricultural land is of particular importance. Relatedly, a need exists to determine the availability of the information required to assess the problem.

1.2 Summary of Findings and Conclusions

The information on nutrient transport from agricultural lands was evaluated to obtain estimates of nutrient input rates to the Great Lakes per unit of land according to crop use category. Data based on runoff from small field plots gave usable information on soluble nutrient transport but overestimated total nutrient transport as a large portion of the eroded soil would not reach the Great Lakes. However, this data allowed comparison of various crop use categories. The most reliable data for estimating nutrient transport to lakes was based on streams draining primarily agricultural areas. However, this data varied considerably among investigations and did not allow evaluation of crop use categories. Based on streams draining agricultural lands, the phosphorus loading was estimated to be about 0.4 kg per hectare (0.35 lb per acre) for total P and 5 kg per hectare (4.5 lb per acre) for total N. For phosphorus, the estimated contribution from agricultural land was about 20 percent of the total phosphorus loading to the Great Lakes. Total nitrogen loadings for the Great Lakes were not available. Because of the range in the data used, the above estimates may be inaccurate.

Within the range of available data, differences in transport rates for soluble nutrients were not found for the different crop use categories. Total nutrient transport from field plots was related to crop cover and corresponding soil erosion. Estimated nutrient loadings were related more to land area than differences in input rates for different crop categories.

The area devoted to agricultural land in the Great Lakes Basin will not increase appreciably in the future. Consequently, any increase in nutrient transport to the Great Lakes from agricultural land will apparently result only if more intensive agriculture increases nutrient losses. In this regard, the effect of fertilization is important. At present, quantitative analysis of the effect of fertilizer use on a basin-wide basis can not be made.

Research and demonstration projects are needed to more accurately assess the problem and establish adequate control measures. Monitoring of representative agricultural drainage basins is needed to provide quantitative information on nutrient transport as related to land use and other important factors. Research is needed on the factors and mechanisms controlling nutrient transport from agricultural lands, especially for soluble nutrients, to determine the role of soil properties, fertilizer use and other factors in controlling nutrient concentrations in runoff. Transport in streams and rivers should be evaluated to provide information on losses due to interaction with stream sediments. The effects of particulate nutrients on the nutrient status of lake surface waters should be determined to clarify the importance of including particulate nutrient forms in lake nutrient budgets. Demonstration projects are also needed to establish the effectiveness of control measures, such as soil conservation and erosion control programs, and the feasibility of implementing control policies and programs.

Section 2

SUPPORTING MATERIAL

2.1 Introduction

Eutrophication is regarded as one of the most important water quality problems for the Great Lakes. Phosphorus and/or nitrogen are considered to be the nutrients limiting the growth of algae in most lake waters, and control of these nutrients has been emphasized as the approach for controlling eutrophication. In particular, phosphorus reduction has been selected as the method of control because reduction of the phosphorus loading is considered more feasible than nitrogen control. Progress has been made in reducing the phosphorus loading through treatment of municipal waters.

The large area of agricultural land in the Great Lakes Basin raises concern over the nutrient load contributed by agricultural lands as well as other diffuse sources. Even a relatively small input per unit land area could result in a relatively large nutrient input. Consequently, a need exists to evaluate the impact of agricultural land on the nutrient loading of the Great Lakes. Among the goals of this review were to evaluate the nutrient loading to the Great Lakes from agricultural lands, and to determine the availability of information needed to conduct this evaluation.

2.2 Scope of Study

An attempt was made to gather substantially the relevant information available relating to nutrient transport from agricultural lands. Information was obtained from scientific publications, project reports, and state and federal reports (see Section 7). The Great Lakes Basin Framework Study was an important source of information on the Basin. The report of the Great Lakes Water Quality Board (1973) provided information on the Great Lakes, particularly estimates of total phosphorus loadings.

Information available to the present time was utilized, including information on ongoing research when it could be obtained.

2.3 Study Procedure

Information on nutrient transport from agricultural land was gathered and evaluated with the ultimate goal of providing information on the contribution of agricultural lands to the nutrient loading of the Great Lakes. An intensive survey of the scientific literature was conducted. Reports from research and demonstration projects were utilized. Letters were sent to individuals and organizations throughout the country thought to have relevant information. These letters requested publications and other information on the subject under investigation. University and state experts were consulted. Information on research in progress was obtained from Smithsonian Science Information Exchange.

The various assumptions made are discussed in the appropriate sections of the review. The assumption of general importance made was that the only nutrients of importance regarding effects on the Great Lakes were phosphorus and nitrogen.

Section 3

REVIEW OF FINDINGS OF PREVIOUS STUDIES

3.1 General Description Of Land Use Category

The category on Nutrients - Agricultural Land is concerned with agricultural land use - water quality relationships. The primary focus is on nutrient loadings to the Great Lakes from agricultural lands as related to type of agricultural land use. The categories of agricultural land use considered are a) row crops, b) close-grown crops, c) pasture and meadows, d) orchards and vineyards, and e) idle cropland.

3.2 Length of Time Activities in Practice

Land settlement of the Great Lakes Area began about 1820. Except for small cultivated areas, the region was nearly all forested. A major change in land use came with land settlement, land clearing, and shifting of much of the land into farms (EPA, 1971). By 1850, 1.8 million hectares (4.4 million acres) were in farms with 0.8 million hectares (1.9 million acres) in improved farm land. The area of improved farm land had increased to 5.3 million hectares (13 million acres) by 1920. In 1966, the total cropping area, including idle cropland, was about 11.6 million hectares (28.6 million acres). Land clearing and farming resulted in increased runoff, soil erosion, and nutrient loading to surface waters from land within the region. Although modern farming and soil conservation practices reduce the runoff and erosion problem, concern exists over whether current agricultural practices are adequate to protect water quality in the Great Lakes.

3.3 Types and Nature of Nutrients Associated With Agricultural Runoff

The category on Nutrients - Agricultural Land is concerned with nutrients considered to be of importance to the water quality of the

Great Lakes. Phosphorus and/or nitrogen have been implicated as the nutrients limiting algal growth in various lakes (Vollenweider, 1968). In the Great Lakes, the major focus has been on phosphorus, due in part to the greater possibility of phosphorus control. Various other nutrients essential to plant growth are contained in agricultural runoff, but there is little evidence that nutrients other than nitrogen and phosphorus limit algal growth in lake waters.

Phosphorus and nitrogen occur in various chemical and physical forms which affect their mobility in agricultural runoff. These relationships are discussed in sections 3.5 and 3.6 of this review.

3.4 State-of-the-Art In Assessing and Quantifying Nutrient Loadings to the Great Lakes from Agricultural Lands

3.4.1 Nutrient Contributions from Agricultural Lands[1]

The transport of nutrients from agricultural lands to the Great Lakes could occur along innumerable pathways and could involve many transport mechanisms. For purposes of this report, water is considered to be the primary transport vector, although it is recognized that nutrient loss by wind-blown particulate matter could be large in some instances.

Three data grouping were prepared which describe the flux of nitrogen and phosphorus from agricultural lands 1) seepage studies, 2) runoff studies and 3) drainage area studies. The first two study types refer to the transport of nutrients across the boundaries of land areas. The latter type refers to nutrient transport by continually flowing streams. It is important to recognize that only a few of the investigations cited were conducted to quantify nutrient runoff from watersheds to lakes. The objective of most studies was to measure nutrient loss from distinct land parcels. Therefore, questions of subsequent nutrient transport must be addressed before the data can be used for estimating nutrient contributions

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[1] This section is based on the report by Uttormark, Chapin and Green (1974).

from agricultural lands.

Seepage studies include lysimeter work and analyses of tile-drained fields. Data from these investigations may be useful in estimating the transport of nutrients from surface soils to groundwater aquifers, but the applicability of these results to the estimation of lake loadings depends greatly on the extent of groundwater-surface water exchange in the basin under consideration.

Losses of nutrients by storm runoff have been quantified in a number of studies. However, the ability to predict nutrient loadings from these data is again limited by the difficulty in defining the probability to transport from agricultural lands to lakes. This difficulty is amplified by the fact that a large portion of the nutrients lost from agricultural lands are associated with particulate matter which may settle out at intermediate points along the flow path, especially during overland flow or in intermittent stream channels.

From the standpoint of lake loadings, some of the most useful data are provided by drainage area studies because of the clearly defined pathway for nutrients contained in continually-flowing streams. However, some trade-offs are involved. Land use patterns are defined clearly for most seepage and runoff studies, but subsequent transport of nutrients to a lake is questionable. In contrast, nutrient flow paths are more clearly defined for drainage area studies, but descriptions of land use are very imprecise (i.e., "typical rural environment" or "mixed farmlands and woodlots").

3.4.1.1 Nutrient Losses by Seepage

Nutrient loss from agricultural lands by seepage through the soil profile has been a topic of interest for several decades. The majority of the earlier studies were conducted to evaluate alternative agricultural practices, such as crop rotation, fertilizer usage, plowing techniques, etc., and losses of nitrate nitrogen were studied almost exclusively. In more recent years, emphasis has been placed on evaluating the effects of agricultural practices on water quality, and losses of phosphorus as well as nitrogen have been reported.

Two types of studies are reported in the literature which describe the transport of nutrients by seepage through soils. These involve the use of lysimeters or the analysis of waters which flow from tile-drained croplands.

Lysimeters are constructed by surrounding a volume of earth on the sides and bottom with an impermeable material. The enclosed soil is more or less disturbed, depending on how it is placed in the enclosure. The top is exposed to the elements, and drains are connected to the bottom, so that all water percolating through the soil profile can be collected and analyzed. Runoff and erosion are generally prevented by the design of the unit. Surface areas of these units range from 0.1 to 10m^2 (0.009 to 0.9 ft^2), and it is reported that some units have been maintained actively for periods as long as 35 years.

In some parts of the country, particularly those regions in which irrigation is practiced, large tracts of land are underlain with tiles or other subsurface collection systems. Some of the irrigation water percolates through the soil profile and is removed by these drainage systems. A number of studies have been reported in which these drainage waters were sampled periodically and analyzed for nitrogen and phosphorus to measure nutrient losses. In many respects, studies of this type are similar to lysimeter studies, but the surface area covered are much larger, and not all of the seepage water is collected--some is lost to deeper aquifers.

The data from seepage studies presented in this section are grouped according to crop, but crop cover plays only a partial (and possibly minor) role in influencing nutrient loss.

Bolton, Aylesworth, and Hore (1970) measured nutrient losses in tile drainage effluent from twelve 0.1 ha (0.25 ac) plots at Woodslee, Ontario (clay soils). Seepage flows were recorded continuously, and effluent samples were filtered to remove sediment and analyzed for total nitrogen and phosphorus. The authors concluded that the nutrient loss was influenced predominantly by the amount of water that percolated through the soil.

Crop	Nutrient loss (kg/ha/yr)* N No fert.	N Fert.	P No fert.	P Fert.
Rotation:				
Corn	5.6	15.1	0.13	0.24
Oats and alfalfa	4.3	5.7	0.13	0.13
Alfalfa, 1st year	4.8	3.9	0.13	0.15
Alfalfa, 2nd year	4.7	8.6	0.08	0.22
Continuous:				
Corn	6.6	14.0	0.26	0.29
Blue grass sod	0.3	0.7	0.01	0.12
Mean:	4.4	8.1	0.12	0.19

*kg/ha/yr x 0.89 = lb/ac/yr

In contrast, Sylvester and Seabloom (1962) monitored irrigation return flows in the Yakima Valley in Washington and found that more nitrate and soluble P were lost during the 6-month non-irrigation period than during the irrigation season even though 130 cm (51 inches) of water were applied (average rainfall was 18 cm (7 inches)/yr). The return flow consisted of both surface runoff and seepage through subsurface drains which continued to flow during the non-irrigation season. Samples were collected and analyzed for nitrate, total Kjeldahl nitrogen, and dissolved and total phosphorus. This project was also

	Nutrient loss (kg/ha)* Irrigation season	Non-irrigation season	Annual
Nitrate - N	34.	39.	73.
Dissolved - P	0.78	1.7	2.5

*kg/ha x 0.89 = lb/ac

described by Sylvester (1961) and separate results are given for surface and subsurface drains.

	Nutrient loss (kg/ha/yr)*			
	Total - N		Total - P	
	Range	Mean	Range	Mean
Surface drains	2.8-27	(16)	1.0-4.4	(2.5)
Subsurface drains	61-186	(103)	3.8-10	(7.7)

*kg/ha/yr x 0.89 = lb/ac/yr

Studies in which nutrient losses from croplands by tile drainage were compared to losses via surface runoff were also conducted in California and Idaho.

Johnston, Ittihadich, Daum and Pillsbury (1965) analyzed tile drainage effluent and surface runoff from irrigated land in the San Joaquin Valley of California. Four 19 to 60 ha (47 to 148 ac) plots growing cotton, alfalfa and rice were studied. The soils were heavy silty clays, and tile depth averaged 190 cm (75 inches). Analyses were made for total nitrogen and total phosphorus. No estimates were made as to the quantity of nitrogen or phosphorus lost through deep percolation.

	Nutrient loss (kg/ha/yr)*		Fertilizer applied (kg/ha/yr)*	
Crop	Total - N	Total - P	N	P
Cotton and rice			300	52
Tile drainage	110.	0.19		
Surface runoff	11.	0.81		
Cotton			220	36
Tile drainage	13.	0.05		
Surface runoff	6.4	0.17		
Alfalfa			none	none
Tile drainage	3.5	0.08		
Surface runoff	1.6	0.20		
Rice			94	none
Tile drainage	42.	0.60		
Surface runoff	5.2	0.11		

*kg/ha/yr x 0.89 = lb/ac/yr

Carter, Bondurant, and Robbins (1971) measured nitrate and dissolved phosphorus in surface runoff and subsurface (tile) drainage from an 82,000 ha (202,600 ac) tract in southern Idaho. The major crops were alfalfa, beans, grain, sugarbeets, corn, and pasture. Precipitation averaged 21 cm (8.3 inches) per year and irrigation supplied an additional 200 cm (79 inches) of water annually.

Nutrient inputs in kg/ha/yr*	N	P
Fertilizer	60.	30.
Irrigation water	2.3	1.0
Precipitation	negligible	negligible
Nutrient losses in kg/ha/yr*	**NO_3-N**	**Sol-P**
Surface runoff	0.35	0.17
Tile drainage	35.	0.13
Total loss	35.	0.30

*kg/ha/yr x 0.89 = lb/ac/yr

In each of these studies, seepage losses of nitrogen were large compared to losses by surface runoff. On the other hand, phosphorus losses through surface runoff tended to be larger. However, phosphorus losses through seepage were sufficiently large to be of significance from a water quality standpoint.

Erickson and Ellis (1971) measured the nutrient content in drainage waters from three experimental farms in Michigan, and compared the nutrient losses to the amounts of fertilizer used.

	Nutrients added (kg/ha/yr)*		Nutrients lost (kg/ha/yr)*		Lost/added (percent)	
	N	P	NO_3-N	Tot-P	N	P
Ferden Farm	90	39	12	0.10	13	0.2
Davis Farm	39	50	8	0.09	20	0.2
Muck Farm	56	17	19	1.5	(34)	(8.8)

*kg/ha/yr x 0.89 = lb/ac/yr

Analyses of seepage from uncultivated, unfertilized land adjacent to the Muck Farm led to the conclusion that the high values of nutrient loss were in part due to accretion from surrounding lands, and that only a small part of the nitrogen and phosphorus added to the Muck Farm reached the drainage water.

Losses of nitrogen as a function of fertilizer usage was also studied by Broadbent and Chapman (1950). They grew vetch, clover and mustard in lysimeters at Riverside, California. The experiment covered a 15-year period and the average water application (rainfall plus irrigation) was 89 cm (39 inches).

	Fertilizer N applied (kg/ha/yr)*		
	0	112	224
	Nitrogen loss (kg/ha/yr)*		
Crop:			
Vetch	30	79	100
Clover	39	63	91
Mustard	20	34	45
Average loss for all crops and treatments:		56	

*kg/ha/yr x 0.89 = lb/ac/yr

Allison et al (1959) reported the results of experiments conducted near Columbia, South Carolina, in which crotalaria, millet, rye, cowpeas, and corn were grown in lysimeters. The lysimeters were 1.2 m(3.9 ft) deep and were filled with sandy soil. Fertilization was reported to be "low." The average annual rainfall was 108 cm (42.5 inches) during the 12-year period covered by the study.

	Nutrient loss (kg/ha/yr)*					
	NO_3-N		Total - N		Total - P	
	Range	Mean	Range	Mean	Range	Mean
Various crops	2.4-40	24	3.6-46	29	0.09-0.17	0.11
Fallow or crops returned to soil			38-140	90		
No crop:						
Fertilized		40		44		0.20
No Fertilizer		30		34		0.14

*kg/ha/yr x 0.89 = lb/ac/yr

Dreibelbis (1946) also found a correlation between nitrate loss and fertilizer usage, but Hendrick and Welsh (1938) reprted no significant differences in nitrate loss between fertilized and nonfertilized plots in a ten-year study conducted in England.

A summary of the data giving nutrient losses from croplands by seepage through the soil profile is given in Table 1. Based on considerations of nutrient pathways, data from lysimeter or subsurface drainage studies are probably most applicable for estimating nutrient loadings of lakes which receive irrigation return waters. The data may also be useful for estimating the flux of nutrients from croplands to groundwater aquifers, but subsequent transport would be highly speculative in most instances.

3.4.1.2 Nutrient Transport From Agricultural Lands In Surface Runoff

A separate data grouping was prepared for nutrient losses from agricultural lands by surface runoff. In studies of this type, samples of runoff water, including suspended matter, are collected periodically from fields or experimental plots. Runoff is not continuous, but occurs only when excessive water is applied through irrigation or rainfall.

Table 1 Nutrient Export From Croplands by Seepage Through Soil Profile

Crop--study	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	References
Corn--tile drainage, Lithuania	2.6					Kinderis (1970)
Corn-oats-hay rotation--lysimeter, 120 kg+ N/ha added, New York	43.					Bizzell (1944)
Corn-oats-hay rotation--lysimeter, New York	2.4					Bizzell & Lyon (1928)
Corn-oats-wheat-hay rotation--lysimeter, New York	5.4 with legumes 7.4 without legumes					" " "
Corn--lysimeter, Ohio	1.9					Dreibelbis (1946)
Corn--tile drainage, Ontario						
No fertilizer			6.1	0.20[a]		Bolton et al. (1970)
Fertilizer added			14.	0.26[a]		" " " "
Cotton--tile drainage, 280 kg N/ha added, California	4.1[b]					Meek et al. (1969)
Cotton--tile drainge, 220 kg N/ha and 36 kg P/ha added, California			13.		0.05	Johnston et al. (1965)
Cotton & rice--tile drainage, 300 kg N/ha and 52 kg P/ha added, California			110.		0.19	" " " "
Rice--tile drainage, 94 kg N/ha added, California			42.		0.60	" " " "

Table 1 (Continued) Nutrient Export From Croplands by Seepage Through Soil Profile

Crop--study	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	References
Oats--lysimeter, England	2.5					Hendrick & Welsh (1938)
Barley--lysimeter, England	8.					" " "
Wheat--lysimeter, Ohio	3.					Dreibelbis (1946)
Hay--lysimeter, England	9.					Hendrick & Welsh (1938)
Timothy--lysimeter, 140 kg N/ha added, New York	12.					Bizzell (1944)
Alfalfa--tile drainage, California			3.5		0.08	Johnston et al. (1965)
Alfalfa--tile drainage, Ontario			4.8	0.1		Bolton et al. (1970)
Alfalfa--lysimeter, Kentucky	11.					Karraker et al. (1950)
Lespedeza--lysimeter, Kentucky	65.					" " " "
Lespedeza & rye--lysimeter, Kentucky	17.					" " " "
Lespedeza & bluegrass--lysimeter, Kentucky	22.					" " " "
Legumes--tile drainage, Lithuania	1.5					Kinderis (1970)
Vetch--lysimeter, California			30.			Broadbent & Chapman (1950)
Clover--lysimeter, California			39.			" " "

Table 1 (Continued) Nutrient Export From Croplands by Seepage Through Soil Profile

Crop--study	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	References
Mustard- -lysimeter, California			20.			Broadbent & Chapman (1950)
Grasses--lysimeter, New York	2.8					Bizzell & Lyon (1928)
Grasses--tile drainage, Lithuania	0.3					Kinderis (1970)
Grasses & wheat--tile drainage, Lithuania	0.8					" "
Bluegrass sod--tile drainage, Ontario			0.3	0.01[a]		Bolton et al. (1970)
Meadow--lysimeter, Ohio	4.3					Driebelbis (1946)
Pasture--lysimeter, England	2.6					Hendrick & Welsh (1938)
Various crops--tile drainage, Idaho	35.			0.13		Carter et al. (1971)
Various crops--tile drainage, Washington	73.		100.	2.5	7.7	Sylvester & Seabloom (1962)
Not stated--tile drainge, Illinois	18.					Harmeson (1971)
Not stated--tile drainage, Michigan, Ferden Farm			12.		0.10	Erickson & Ellis (1971)
Davis Farm			8.		0.09	" " "
Muck Farm			19.		1.5	" " "

Table 1 (Continued) Nutrient Export From Croplands by Seepage Through Soil Profile

Crop--study	N (kg/ha/yr)*			P (kg/ha/yr)*		References
	NO_3	NH_4	Total	Dissolved inorganic	Total	
Fallow (no crop)--lysimeter, South Carolina	30.				0.14	Allison et al. (1959)
Fallow--lysimeter, New York	76.					Bizzell & Lyon (1928)
Fallow--lysimeter, Kentucky	82.					Karraker et al. (1950)

[a] Total dissolved P

[b] $NO_3 + NO_2$ for 8 mo.

*kg/h a/yr x 0.89 = lb/ac/yr

+kg x 2.2 = lb

As was the case for seepage studies, most surface runoff investigations were conducted to evaluate alternative farming practices, such as plowing techniques, crop rotations and fertilizer applications, from the standpoint of minimizing soil and nutrient losses from croplands. Consequently, particulate material was intentionally included in the samples, and particulate, along with dissolved, nutrients were measured in most instances. Study areas were often quite small and usually the areas were devoted to single crops. Fertilization and plowing were generally uniform within a study area, but large differences occurred between study areas.

Data from surface runoff studies are given in Table 2. The data are grouped according to crop, but other factors, such as slope, soil characteristics, farming practices, and antecedent soil moisture, as well as duration, frequency and intensity of precipitation, may have a greater influence on the quantity of nutrients lost from croplands. Slopes (not always given) ranged from 3-20% for the studies reported, and the annual amounts of rainfall and irrigation water ranged from 75 to 220 cm (30 to 87 inches).

Data from surface runoff studies may be useful for estimating nutrient inputs from agricultural lands immediately adjacent to lakes. However, as shown in Table 2, but far the largest amount of nitrogen and most of the phosphorus lost from croplands was associated with particulate matter. The likelihood that particulate matter will be transported sufficiently far to enter a lake must be taken into account before surface runoff data are used to estimate nutrient contributions from croplands which exist in distant portions of a watershed.

3.4.1.3 Nutrient Transport by Streams Draining Agricultural Lands

A number of studies are reported in the literature in which streams draining predominantly agricultural watershed were monitored for nutrient content. These studies typically involved the continuous measurement of streamflow and periodic sampling for nutrient determinations.

Table 2 Phosphorus and Nitrogen in Runoff from Crop Land

Land Use Practices			Solution			Particulate		Total		Reference
Crop	Fertilization	Management	Pi	NO_3-N	NH_4-N	Tot-P	Tot-N	P	N	
			kg/ha/yr*							
Row Crops										
Corn	M	G						1.90	4.4	Minshall et al. (1970)
"	H	P	0.07	0.80		0.11	12.6	0.18[2]	13.4[2]	Timmons et al. (1968)
"	H	P	0.70	1.20		0.04	4.2	0.74[2]	5.4[2]	Timmons et al. (1968)
"	M	P						1.50	23.0	Massey et al. (1953)
"	H	G	0.17[1]	1.70	1.30	1.05	36.6	1.22[2]	39.6[2]	Schuman et al. (1973)
"	M	G	0.11	1.00	0.90	0.58	23.1	0.69[2]	25.0[2]	Schuman et al. (1973)
"	H	G	0.05	0.20	0.30	0.08	2.6	0.13[2]	3.1[2]	Schuman et al. (1973)
"	H	G	0.13	1.41	0.33					Klausner et al. (1974)
"	H	P	0.49	2.46	0.99					Klausner et al. (1974)
"	M	G	0.05	0.39	0.14					Klausner et al. (1974)
"	M	P	0.16	0.40	0.21					Klausner et al. (1974)
"	M	P						6.60	182	Weidner et al. (1969)
Beans	H	G	0.18	1.46	0.40					Klausner et al. (1974)
Idle Crop Land										
Follow	H	-	0.05[1]	3.9	0.34		63	0.39[2]	66.9[2]	Timmons et al. (1968)
"	None	-			2.07					White & Williamson (1973)
Orchard and Vineyards										
Apple	M	G						1.40	0.9	Weidner et al. (1969)

Table 2 (Cont.) Phosphorus and Nitrogen in Runoff from Crop Land

Land Use Practices			Solution			Particulate		Total		Reference
Crop	Fertilization	Management	Pi	NO_3-N	NH_4-N	Tot-P	Tot-N	P	N	
			kg/ha/yr*							
Close-Grown Crops										
Oats	H	P	0.01[1]	0.75		0.03	5.2	0.04[2]	6.0[2]	Timmons et al. (1968)
"	M	G				0.32				White & Williamson (1973)
Wheat	M	G	0.04	2.46	0.21					Klausner et al. (1974)
"	M	P	0.09	2.29	0.61					Klausner et al. (1974)
"	H	G	0.17	0.93	0.42					Klausner et al. (1974)
"	H	P	0.32	1.14	0.36					Klausner et al. (1974)
"	M	G	0.21	0.50	1.30					Klausner et al. (1974)
"	M	P	0.08	0.60	0.15					Klausner et al. (1974)
"	M	P						0.90	24	Weidner et al. (1969)
Pastures and Meadows										
Hay	H	P	0.23[1]	3.5					3.5	Timmons et al. (1968)
Alfalfa	H	-						0.20	1.6	Johnston et al. (1965)
"	None	-				0.07				White & Williamson (1973)
Bromegrass	M	G	0.21	0.70	0.40	0.07	1.3	0.28[2]	2.4[2]	Schuman et al. (1973)

*kg/ha/yr x 0.89 = lb/ac/yr

[1]Total P in solution

[2]Total obtained as sum of inorganic nutrients in solution and particulate nutrient concentrations.

In many cases, sampling frequency was related to streamflow so that additional samples were collected during periods of high flow. The amounts of nutrients transported by the streams were then calculated from a streamflow record and a time series of nutrient concentrations. Two somewhat different approaches were used to accomplish this: 1) The flow hydrographs were subdivided into segments (usually centered about the dates on which water samples were collected), and a single nutrient concentration was then assumed to be characteristic of the total water mass passing the gaging station during each time segment; 2) sample analyses were used to develop concentration-stream-flow realtionships, and nutrient transport was computed by applying these realtionships to the stream hydrograph and integrating over appropriate time intervals.

A summary of nutrient transport from agricultural lands by steramflow is given in Table 3. Flux coefficients for these studies are less variable than for seepage or surface runoff studies, but reported values still cover a considerable range as shown below.

	Total N	Total P
	kg/ha/yr(lb/ac/yr)	
maximum	13.0(11.6)	2.3(2.0)
minimum	1.2(1.1)	0.03(0.03)
average	5.1(4.5)	0.38(0.34)

Of the 24 values of total-phosphorus that were compiled, only 7 were larger than 0.4 kg-P/ha/yr (0.36 lb/ac/yr), and 6 of the 7 were from the Midwest (2 from Illinois, 4 from Wisconsin). The largest value reported, 2.3 kg/ha/yr (2.05 lb/ac/yr), was determined from a study in Arkansas conducted by Gearheart (1969). In this study, streamflow draining a 3,072 km^2 (1,186 $mile^2$) watershed (80% agricultural, used primarily for pasture and poultry production) tributary to Beaver Reservoir was monitored for a seven-month period from October through April. Estimates of annual values were presented by the author. Whereas the phosphorus value was

Table 3 Nutrient Transport From Agricultural Watersheds By Streams

Location - comment	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	Tot-N / Tot-P	References
Illinois, Kaskaskia River basin							
Subbasin area / % agricultural							
32 km^{2+} / 86.0	11.				0.03[a]		Engelbrecht & Morgan (1959)[d]
320 " / 86.2					0.05[a]		
2,700 " / 81.7					0.85[a]		Harmeson et al. (1971)
5,100 " / 76.0					0.44[a]		
6,900 " / 68.9					0.11[a]		
13,500 " / 76.8					0.10[a]		
Connecticut, 85 km^2 watershed, 50% forested, "typical rural environment"			3.4		0.22	15.	Frink (1967)
Arkansas, 3072 km^2 watershed, 80% agricultural			3.6		2.3	1.5	Gearheart (1969)
Potomac River Basin (Catoctin Creek) 280 km^2, 80% farmland, 20% forest	3.8[b]		4.3		0.27	16.	Jaworski & Hetling (1970) Jaworski et al. (1969)
Ontario, tributaries of Bay of Quinte, 50% agricultural, 50% forests, many lakes and bogs							
River / Area (km^2)							
Trent / 13,000			2.1		0.11	19.	Johnston & Owen (1971)
Moira / 2,700			1.8		0.08	22.	
Salmon) / 1,660			2.4		0.07	34.	
Napanee)			3.0		0.14	21.	
North Carolina, Pigeon River watershed, 350 km^{2+}					0.17		Keup (1968)

Table 3 (Continued) Nutrient Transport From Agricultural Watersheds By Streams

Location - comment	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	Tot-N / Tot-P	References
Maine, rural areas, sparsely populated, Stetson R, 74 km^2			1.9		0.04	48.	Mackenthun et al. (1968)[d]
Wisconsin, average for 36 streams, base flow only, 5.7-370 km^2			1.2		0.10	12.	Minshall et al. (1969)
Ontario, Grand R watershed, 3500 km^2					0.07		Missingham (1967)[d]
Ontario, near Toronto							
River (Area (km^2))							
West Humber 130 (dairy farms)			3.2		0.21	15.	Owen & Johnson (1966)
Little Rouge 78 (mixed farms)			8.4		0.35	24.	Neil, Johnson & Owen (1967)
Antona 54 (mixed farms)			4.0		0.17	24.	
England							
Arable land			13.				Owens (1970)
Permanent pasture			8.				
Wisconsin, tributaries to Lakes:							
Monona	← 4.9	→	6.7	0.06			Sawyer (1947)
Waubesa	← 5.5	→	7.6	0.11	0.44	17.	
Kegonsa	← 7.2	→	9.2	0.11	0.46	20.	
Prince Edward Island, 26 km^{2+} watershed, 28% potato fields, remainder in pasture & woodlot				0.21			Smith (1959)

Table 3 (Continued) Nutrient Transport From Agricultural Watersheds by Streams

Location - comment	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	Tot-N / Tot-P	References
Ohio, 123 ha watershed, 25% woodlots, 50% pasture, 25% cropland (data for 4 consecutive years)	1.1		1.7	0.03			Taylor et al. (1971)
	2.2		3.1	0.08			
	9.1		10.6	0.07			
	1.8		4.4	0.08			
Wisconsin, Menomonee R watershed					1.6[c]		Zanoni (1970)[e]
Wisconsin, 546 ha watershed, dairy farming, 0.15% slopes	3.1[b]	1.3	8.8	0.58	0.77	11.	Zitter (1968)

[a] Ortho-P + maximum inorganic condensed-P - Authors state that total-P values may be 20-30% higher than those reported.

[b] $NO_3 + NO_2$

[c] Total dissolved - Author states that values are within a few percent of total-P.

[d] Data given as loss/day or loss for part of a year--extrapolated linearly to an annual value.

[e] Approximated from data presented.

* kg/ha/yr x 0.89 = lb/ac/yr

+ km^2 = 247 ac.

larger, the flux of nitrogen, 3.6 kg-N/ha/yr (3.2 lb/ac/yr), was less than the average for other data included in this group, and the ratio of Tot-N/Tot-P was only 1.5. Ratios for all other studies were greater than 10.

Phosphorus transport in the Kaskaskia River watershed in Illinois was reported by Engelbrecht and Morgan (1959, 1961). Water samples were collected at approximately weekly intervals at four sites and monthly at three additional locations during the period from April-December 1956. Some municipal effluents were discharged to the river, but contributions from these sources were subtracted from the total load, and values for land drainage are presented by the authors. Results presented as phosphorus loss/unit area/day were extrapolated linearly to annual values here.

Mackenthun, Keup, and Stewart (1968) and Mackenthun (Chairman, 1966) reported studies on the results of tributaries to Lake Sebasticook, Maine. The streams studied drained sparsely populated rural areas with no significant waste discharges. The primary crops were potatoes, apples, alfalfa, beans, and corn which received an average of 82 kg/ha (73 lb/ac) of phosphorus as fertilizer. Precipitation was 102 cm/yr (40 inches/yr). Water samples were collected during one- to two-week periods in February, May, July-August, and October-November. Nutrient losses per day were reported by the authors for each of the four sampling periods.

	Nutrient Loads (kg/ha/day)*			
	Stetson Stream		Mulligan Stream	
	N	P	N	P
Winter	0.0029	0.0	No Flow	
Spring	0.0086	0.00015	.00084	.00003
Summer	0.0083	0.00012	No Flow	
Fall	0.0015	0.00018	.0013	.00005

*kg/ha/day x 0.89 = lb/ac/day

Annual values were computed here by multiplying each daily load per season by 91.25 and summing.

Phosphorus transport from a 3,500 km^2 (1,350 $mile^2$) agricultural watershed in Ontario (Grand River) was repoted by Missingham (1967).

Month	Total P (kg/ha/day)*
December	0.00014
January	0.00025
February	0.00021
Average	0.00020

*kg/ha/day x 0.89 = lb/ac/day

An average value of 0.0002 kg/ha/day (0.00018 lb/ac/day) converts to 0.07 kg/ha/yr (0.062 lb/ac/yr) which is, most likely, an underestimate of the amount of phosphorus transported annually from the basin.

Minshall, Nichols, and Witzel (1969) carried out a two-year study to determine the amount of nutrients in base flow of southwestern Wisconsin streams. Flow rates were determined for 36 streams with drainage areas varying from 570 to 37,000 ha (1,410 to 91,430 ac). Samples were collected, and flows were measured at times when no surface runoff was entering the streams.

The area studied was 90% agricultural, with 40% in contour strip-cropped farmland (corn, oats and alfalfa), 40% in pasture, and 10% woodland. Livestock enterprises were prevalent. Soils were moderately and well-drained silt loams, and the mean annual precipitation for the area was 83 cm (32.7 inches). An average of 9 kg/ha/yr (8 lb/ac/yr) nitrogen was applied as manure or artificial fertilizers.

	Nutrient Loss (kg/ha/yr)*			
	1966		1967	
	Tot-N	Tot-P	Tot-N	Tot-P
High	4.2	0.25	5.5	0.49
Low	0.4	0.01	0.5	0.03
Weighted Average	1.1	0.08	1.4	0.12

*kg/ha/yr x 0.89 = lb/ac/yr

Zanoni (1970) conducted a study of the Menomonee River basin in southeastern Wisconsin, and reported that an average of 1.18 kg/ha (1.05 lb/ac/yr) of total soluble phosphorus was contributed annually to Lake Michigan from land drainage in the watershed. The total watershed contains 350 km^2 (135 $mile^2$) of which 38% is agricultural, 43% is urban and the remaining 19% is woodlots, parks and unproductive land. An analysis of runoff from sub-basins in the watershed showed that urban areas contributed 0.58 kg/ha/yr (0.52 lb/ac/yr). Drainage from the remainder of the watershed, primarily rural lands, can be computed to yield a contribution of 1.63 kg/ha/yr (1.45 lb/ac/yr).

3.4.2 Nutrient Loadings for the Great Lakes

Estimates have been made of the P loadings for the five Great Lakes and of the N loadings for Lake Huron (Zar, 1972; Great Lakes Water Quality Board, 1973). The greater emphasis on P than N in these estimates apparently reflects the focus on P in regulatory programs planned or implemented to reduce the input of nutrients into the Great Lakes (Zar, 1972; Great Lakes Water Quality Board, 1973).

The P loading for Lake Michigan was estimated by the Phosphorus Technical Committee to the Lake Michigan Enforcement Conference (Zar, 1972). These estimates are summarized in Table 4. For direct and indirect point sources of P discharge, the estimates were based on the assumptions of a P concentration of 10 mg/l in the influent of sewage treatment plants, a discharge to domestic waste water of 1.64 kg (3.65 lb) of P per person per year and 454 liters (120 gallons) per person per day, or were calculated from actual effluent data where available. Consequently, the values for point sources reflect mainly municipal rather than industrial wastes. Direct point sources are wastes discharged directly into the lake, while indirect point sources are wastes discharged into tributaries to the lake. Diffuse sources estimates were based in part on soil erosion (0.5 kg P/metric ton (1 lb P/ton) of sediment or 0.9×10^6 kg P/yr (2×10^6 lb P/yr for the lake), and on other sources (decaying leaves and crop residues; dissolved nutrients in runoff). Details on the basis for estimates of the latter sources were not given. However, estimated total input from diffuse sources

ranged from 12. to 30 kg/km^2/yr (1 to 25 lb/mile^2/yr) or 0.5 to 3.2 x 10^6 kg/yr (1 to 7 x 10^6 lb/yr) basin wide.

Estimates of P loadings from other than direct point sources were also made based on measurements of P concentrations and flow for tributaries to Lake Michigan during 1969. The total input from tributaries and direct point sources (7,764 metric tons/yr or 8,560 tons/yr) is approximately equal to the total input based on estimates (explained previously) of indirect point sources and diffuse sources (7,809 metric tons/yr or 8,609 tons/yr). The close agreement between the two estimates tends to support their validity (Table 4).

The P loadings estimated for Lake Michigan (Table 4) indicate that the contribution from diffuse sources (including agricultural lands) is appreciable (about 25% of the total). However, loadings for diffuse sources are variable and more difficult to estimate than for point sources. Furthermore, the specific contribution from agricultural lands was not estimated. In spite of these problems in assessment, it appears that the P contribution from agricultural lands is significant and will increase in proportion as the input from point sources is reduced on the basis of the requirement that all municipalities achieve 80% reduction of total P entering waste water treatment plants that discharge to Lake Michigan or to one of its tributaries (Zar, 1972).

The P loading estimates for the five Great Lakes are shown in Table 5 according to the contributions from tributaries, industrial sources and municipal sources. The values for municipal and industrial sources are based on direct discharges into the lakes, while tributary loading include diffuse sources (including agricultural lands) and point discharges along the tributaries (EPA, 1972; Great Lakes Water Quality Board, 1973)[1].

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[1] Data for nutrient budgets was taken from Section III-C, Water Quality Conditions (Great Lakes Water Quality Board, 1973). Discrepancies exist in some cases between these values and values in Section IV-C, Eutrophication, of the same publication.

Table 4. Phosphorus Loadings to Lake Michigan (Zar; 1969)[1]

Source	Amount	
	metric tons/year	(tons/year)
Direct point sources	1,771	(1,953)
Indirect point sources	4,221	(4,654)
Sub-total	5,993	(6,607)
Diffuse sources	1,816	(2,002)
Total (Based on estimates of sources)	7,809	(8,609)
Tributaries (1969)	5,993	(6,607)
Direct municipal and industrial sources	1,771	(1,953)
Total (Based on measurement of tributary inputs)	7,764	(8,560)

[1]Estimates do not include direct atmospheric fallout and precipitation and combined sewer overflows.

Table 5.

Total Phosphorus Loadings to the Great Lakes From Tributaries, Industrial and Municipal Sources

(Great Lakes Water Quality Board, 1973)

	Superior		Michigan[1]		Huron		Erie		Ontario	
Canada	metric tons/year						(tons/year)			
Tributaries	608	(670)			1,143	(1,259)	934	(1,030)	1,705	(1,880)
Industrial	118	(130)			1	(1)	54	(60)	299	(330)
Municipal	95	(105)			77	(85)	36	(40)	2,413	(2,660)
United States										
Tributaries	980	(1,080)	5,986	(6,600)	1,390	(1,531)	4,185	(4,615)	1,977	(2,180)
Industrial	-	-	-	-	nd	(nd)	209	(230)	-	-
Municipal	363	(400)	1,814[2]	(2,000)	4	(4)	1,900	(2,095)	1,224	(1,350)
	2,164	(2,385)	7,800	(8,600)	2,615	(2,880)	7,318	(8,070)	7,618	(8,400)
Input from other Great Lake(s)	-	-	nd	(nd)	885[3] 231[4]	(975) (255)	9,578[5]	(10,560)	5,505[6]	(6,070)
	2,164	(2,385)	7,800	(8,600)	3,731	(4,110)	16,896	(18,630)	13,123	(14,470)

[1]Based on estimates reported by Zar (1972)

[2]Includes industrial and municipal sources

[3]Lake Superior, including St. Mary's River

[4]Lake Michigan

[5]Lake Huron, Lake St. Clair, and the Detroit River

[6]Lake Erie, including the Niagara River and Welland Canal

Table 6.
Nitrogen Sources of Lake Huron
(Great Lakes Water Quality Board, 1973).

Source	Amount	
	metric tons/year	(tons/year)
Canada		
Tributaries	16870	(18600)
Industrial	ns	(ns)
Municipal	181	(200)
	17,051	(18,800)
United States		
Tributaries	nd	
Industrial		
Municipal		
	17,051	(18,800)
Input from Lake Superior including St. Mary's River	19,410	(21,400)
Input from Lake Michigan	19,682	(21,700)
	56,143	(61,900)

Estimates for the Great Lakes show that diffuse and indirect point sources account for a major part of the P loadings. The proportion of the total P loading contributed by tributaries ranges from about 77% for Lake Michigan to 28% for Lake Ontario. The lower proportions for Ontario and Erie reflect the importance of contributions from the preceding lake in the chain. While the estimates for the Great Lakes show the importance of P contained in tributaries, these values do not show the relative importance of diffuse sources (including agricultural land) and point waste discharges along the tributaries. Municipal waste discharges into tributaries likely account for a large part of the total tributary load. More detailed information is needed to assess the agricultural contribution.

Apparently, loadings of N to the Great Lakes have received considerably less attention than estimates of P loadings. Estimates were reported for the N loading to Lake Huron from the Canadian side by the Great Lakes Water Quality Board (1973) as shown in Table 6. In contrast to the P loading (about 60% from direct municipal and industrial sources for the Canadian side), the N was estimated to arise almost completely from tributary input and from Lakes Superior and Michigan.

3.4.3 Estimation of the Contribution of Agricultural Lands to the Nutrient Loading for the Great Lakes

Information on the amounts of nutrients transported from agricultural land in surface runoff, summarized in section 3.4.1.2, provides a basis for estimating the amounts of nutrients transported to the Great Lakes from agricultural lands. Use of data obtained from small watersheds or field plots allows estimation of nutrient loadings according to the type of agricultural land use. However, it must be recognized that considerable variability exists in data available for a given land use category, e.g., row crops, (see section 3.4.1.2) and that estimates made on this basis are only approximations at best.

The data on rates of nutrient transport in surface runoff (section 3.4.1.2) was averaged to obtain typical nutrient loading rates for each land use category. Values were estimated for inorganic phosphate in solution (soln Pi), total phosphorus, inorganic nitrogen (NH_4-N+NO_3-N) in solution (soln N) and total nitrogen (Tables 7 & 8). Input rates were obtained by averaging the data for each land use category (Table 11, section 3.4.1.2). Data on agricultural land use was taken from the Great Lakes Basin Framework Study, Appendix 13 (1971).

The estimated phosphorus loading to the Great Lakes from agricultural lands was 1830 metric tons/year (2020 tons/year) of inorganic P in solution and 12,000 metric tons/year (13,300 tons/year) for total P. These values correspond to 9 and 60 percent, respectively, of the total P loading for the Great Lakes estimated by the Great Lakes Water Quality Board (1973) and shown in Table 5, section 3.4.2. Loadings for N were 27,800 metric tons/year (30,600 tons/year) for inorganic N in solution and 399,000 metric tons/year (439,000 tons/year) for total N. As discussed below, total N and P values likely are overestimates, as part of the soil eroded from small agricultural plots is deposited as sediment prior to reaching the Great Lakes. Furthermore, some of the particulate N and P transported to the Great Lakes is unavailable and/or settles to the lake bottom before utilization by aquatic plants can occur (see Section 3.5).

The input rates for inorganic P in solution (Table 7) were similar (about 0.2 kg/ha/yr or 0.18 lb/ac/yr) for the different crop use categories, with the exception of idle crop land. Limited data was available for the latter category. These values are partly affected by the amount of data available and the area in which the data was collected. Considering the range in data used to obtain the input rates, the differences between crop use categories are not significantly different for solution P. This suggests that crop use may not be the best approach for classification of agricultural land for estimating nutrient loading. A greater range was obtained in input rates for total P. The differences reflect the influence of crop cover on the land surface on soil erosion. Thus, the highest rate was obtained for row crops. The values for idle crop land and

Table 7. Estimated Annual Loadings of Phosphorus to the Great Lakes by Runoff from Crop Land (U.S. Region only)

Crop	Area	Input Rate		Annual Loadings	
		Soln P_i	Total P	Soln P_i	Total P
	1,000 hectares (1,000 acres)	kg/ha/yr (lb/ac/yr)		metric tons/yr (tons/yr)	
Row Crops	3,949 (9,750)	0.21 (0.19)	1.62 (1.45)	829 (914)	6,400 (7,050)
Close-Grown Crops	1,458 (3,600)	0.13 (0.12)	0.47 (0.42)	190 (209)	685 (755)
Pastures and Meadows	2,719 (6,715)	0.22 (0.19)	0.24 (0.21)	598 (659)	653 (720)
Orchards and Vineyards	243 (600)	0.21[1] (0.19)	1.40 (1.25)	51 (56)	340 (375)
Idle Crop Land	3,219 (7,947)	0.05 (0.04)	1.23 (1.10)	161 (178)	3,960 (4,370)
Total	11,588 (28,612)			1,829 (2,017)	12,038 (13,270)

[1] Assumed to be the same as row crops.

Table 8. Estimated Annual Loadings of Nitrogen to the Great Lakes by Runoff From Crop Land (U.S. Region only)

Crop	Area	Input Rate		Annual Loadings	
		Solution N	Total N	Solution N	Total N
	1,000 hectares (1,000 acres)	kg/ha/yr (lb/ac/yr)		metric tons/yr (tons/yr)	
Row Crops	3,949 (9,750)	1.56 (1.39)	37.0 (33.0)	6,160 (6,780)	146,000 (161,000)
Close-Grown Crops	1,458 (3,600)	1.67 (1.49)	15.0 (13.4)	2,440 (2,680)	21,900 (24,100)
Pastures and Meadows	2,719 (6,715)	2.30 (2.05)	2.50 (2.23)	6,250 (6,880)	6,800 (7,480)
Orchards and Vineyards	243 (600)	1.56[1] (1.39)	37.0[1] (33.0)	379 (417)	8,990 (9,890)
Idle Crop Land	3,219 (7,947)	3.90 (3.48)	66.9 (59.7)	12,600 (13,800)	215,000 (237,000)
Total	11,588 (28,612)			27,829 (30,557)	398,690 (439,470)

[1] Assumed to be the same as row crops.

orchards reflect data obtained from fallow soil and cultivated orchards, respectively. This may not be typical of land in these categories in the Great Lakes Basin.

As with P, input rates for inorganic N in solution were generally similar for the different crop categories and were low compared to total N. The rates for total N reflected the importance of soil erosion on bare soil.

While data for runoff collected from small plots provides information related to agricultural crop cateogry, these data do not reflect the processes, such as sedimentation, which occur during transport of the water to the Great Lakes. On this basis, data on streams draining agricultural lands probably provides a better measure of nutrient transported to lakes. Even using stream data, the calculated total nutrient loadings will over-estimate the available nutrient loadings (see Section 3.5).

Considerable data is available on nutrients in streams draining agricultural lands (see Section 3.4.1.3). This data can be used to estimate typical input rates for agricultural lands but can not be used to estimate differences for the various crop use categories as the streams generally draining areas including several agricultural land uses. The stream data in Table 3 (Section 3.4.1.3) was used to calculate input rates for dissolved inorganic P, total P, dissolved inorganic N, and total N (Table 9). The value for dissolved inorganic P is slightly higher than the value obtained from runoff data, 1,300 metric tons/year (1,400 tons/year as compared to 1,830 metric tons/year (2,020 tons/year). However, the total P loading rate based on stream data, 4,200 metric tons/year (4,600 tons/year), was considerably lower than the loading rate based on runoff data, 12,000 metric tons/year (13,300 tons/year). The agricultural land loading to the Great Lakes, based on data from streams draining agricultural land is 21 percent of the total loading (Table 9) and represents a better estimate than the value based on runoff.

In the case of N, stream data gives similar loading rates for inorganic N in solution and total N. This reflects the high proportion of

Table 9. Estimated Annual Loadings of Nutrients to the Great Lakes based on Stream data (U.S. Region Only)

Total Cropping Area = $11,588 \times 10^3$ hectares
($28,612 \times 10^3$ acres)

Nutrients	Input Rate	Annual Loadings	Proportion of Loading to the Great Lakes	
			Tributary P	Total P
	kg/ha/yr (lb/ac/yr)	metric tons/yr (tons/year)	%	
Phosphorus				
Dissolved Inorganic P	0.11 (0.10)	1,280 (1,400)	9	6
Total P	0.36 (0.32)	4,170 (4,590)	29	21
Nitrogen				
Inorganic N	3.80 (3.39)	44,000 (48,400)		
Total N	4.90 (4.38)	56,800 (62,500)		

Table 10. Comparison of Phosphorus Sources for Lake Erie and Lake Ontario as Percent Contribution

Inputs	Lake Erie[1]	Lake Ontario[2]
	%	
Direct Precipitation	5	-
Land Runoff		9
Urban	4	
Agricultural or Rural	15	
Forest	-	
Wastewater		
Municipal	73	86
Industrial(Direct)	4	5

[1]Weibel, 1969. Excludes input from Lake Huron which was 12.6% of the total input.

[2]EPA, 1971. Excludes input from Lake Erie.

NO_3-N in the data used (Table 3). As with P, the total N loading rate is considerably lower when based on stream data rather than on runoff data. The total N loading from agricultural land of 56,800 metric tons/year (62,500 tons/year) based on stream data is considered a more reliable estimate than the value of 399,000 metric tons/year (439,000 tons/year) based on runoff. Insufficient information is available on the total N loading to the Great Lakes to compute the proportion arising from agricultural lands.

The proportion of the P loading to the Great Lakes estimated to originate from agricultural lands (21%) can be compared to estimates obtained by others using different approaches. In the case of Lake Michigan (see Table 4), it was estimated that about 23% of the total loading was from diffuse sources. This would include urban runoff and other non-point sources. Out of the total P loadings to Lake Erie, about 14% was estimated to come from agricultural land, while 9% was contributed to Lake Ontario by runoff which also included urban runoff (Table 10).

In summary, agricultural land is estimated to contribute about 20 percent of the total P loading to the Great Lakes and about 30 percent of that contributed by tributaries to the Great Lakes. This proportion will increase as treatment of municipal wastes is accomplished. The largest proportion likely arises from row crops, but appreciable amounts originate from other crop cateogries. Within the present accuracy of loading rates for different land use categories, the differences in loading rates are primarily a function of the land areas in the respective categories rather than differences in loading rates.

3.5 Mobility of N and P within Soil-Water, Ground Water and Surface Water Systems.[1]

[1] The parts of this section dealing with P are based on the review by Ryden *et al.* (1973).

3.5.1 Introduction

The transport of P and N from agricultural lands to surface waters is a function of the physical and chemical factors controlling the mobility of P and N in soils. An understanding of these factors is essential to evaluating the extent of P and N transport in surface, subsurface and ground water runoff as related to land use activities and to developing management programs to minimize nutrient transport.

This section deals with the factors controlling P and N mobility in soil-water systems. Quantitative aspects P and N transport in runoff waters are considered in section 3.4.

3.5.2 Terminology

3.5.2.1 Hydrology and P and N Sources

"Watershed" (drainage basin, catchment area) -- A part of the surface of the earth that is occupied by a drainage system, which consists of a surface stream, or a body of standing (impounded) surface water, together with all tributary surface streams and bodies of standing surface water.

"Stream" -- A general term for a body of flowing water. In hydrology the term is usually applied to the water flowing in a natural channel.

"Stream flow" -- The discharge (of water) that occurs in a natural channel.

"Runoff" -- That part of precipitation that falls on land and ultimately appears in surface streams and lakes. Runoff may be classified further according to its source.

"Surface runoff" (overland flow) -- The part of rainwater or snowmelt which flows over the land surface to stream channels. Surface runoff may also enter standing waters directly or be consolidated into

·tificial channels, e.g., storm sewers in urban areas ("urban runoff"), :fore entering a stream or body of standing water.

"Subsurface runoff" (storm seepage) -- That part of precipitation ıich infiltrates the surface soil and moves towards streams as ephemeral, ıallow, perched ground water above the main ground-water level. In many gricultural areas subsurface runoff may be intercepted by artificial rainage systems, e.g., tile drains, accelerating its movement to streams.

"Ground water runoff" (base runoff) -- That part of precipitation hat has passed into the ground, has become ground water, and is subsequently ischarged into a stream channel or lake as spring or seepage water.

In addition to runoff the other potential contributors to ;treams and standing waters are precipitation incident on the water ;urface and industrial and sewage effluents.

McCarty (1967) and Vollenweider (1968) have made a useful division of sources of P to surface waters based on the ease of quantification. Point sources enter at discrete and identifiable locations and are therefore amenable to direct quantification and measurement of their impact on the receiving water. Major point sources include effluents from industrial and sewage-treatment plants. Diffuse sources may be defined as those which at present can only be partially estimated on a quantitative basis and which are probably only amenable to attenuation rather than elimination. Diffuse sources require the most investigative attention. Vollenweider (1968) further divided diffuse sources into:

i) Natural sources such as eolian loading, and eroded material from virgin lands, mountains and forests.
ii) Artificial or semi-artificial sources which are directly related to human activities such as fertilizers, eroded soil materials from agricultural and urban areas, and wastes from intensive animal rearing operations.

The loads of P imparted to runoff and streams from natural diffuse sources provide a datum line against which the magniture of P loads from artificial sources may be compared.

3.5.2.2 Forms of P and N

In natural systems, P occurs as the orthophosphate anion (PO_4^{3-}) which may exist in purely inorganic form ($H_2PO_4^-$ and HPO_4^{2-}) or be incorporated into an organic species (organic P). Under certain circumstances inorganic orthophosphate may exist as a poly- or condensed phosphate. A secondary distinction is made between particulate and dissolved forms of P, the split conventionally being made at 0.45 μm.

Other terminology used is as follows:

"Total P" -- all forms of P in a runoff or stream sample (dissolved and particulates in suspension) as measured by an acid-oxidation treatment (e.g., acid ammonium persulfate).

"Dissolved inorganic P" -- P in the filtrate after 0.45 μm separation determined by an analytical procedure for inorganic orthophosphate.

"Organic P" -- may be determined within the dissolved and particulate fractions by the difference between the total P and inorganic P.

In soils and waters, N occurs as the nitrate ion (NO_3^-), the ammonium ion (NH_4^+) and in organic compounds (amino N, heterocyclic N, etc.). Small amounts of nitrite (NO_2^-) may be present but the amounts are usually small compared to other forms and NO_2^- is generally not measured in runoff. All of these forms may occur in the dissolved or particulate fraction. The conventional division between the dissolved and particulate fractions is based on filtration using a 0.45 μm pore size filter.

"Total N" - all forms of N in the sample (dissolved and particulate), including NH_4-N, NO_3-N and organic N.

"Dissolved inorganic N" - the sum of NO_3-N and NH_4-N in the filtrate after 0.45 μ m separation. NO_3-N and NH_4-N are determined by separate methods and may be expressed as such. NO_3-N is associated mainly with the dissolved phase and the necessity of removing fine particulates to avoid overestimations is less critical than with P and NH_4-N.

"Organic N" - is determined as the NH_4-N formed on Kjeldahl digestion of the dissolved or particulate fraction.

3.5.3 Physical Factors Affecting the Mobility of P and N

All terrestrially-derived diffuse sources of N and P are associated with the movement of water in contact with a solid phase. The solid phase may be stationary with respect to water flow, or may move in the flow at some speed equal to or less than the flow. Precipitation disposed of as subsurface or ground water runoff is primarily in contact with a stationary solid phase, namely the soil profile and, in the case of ground-water runoff, possibly the bed rock. Consequently, the amounts and concentrations of N and P carried in subsurface and ground-water runoff will be influenced by the time of contact with any component in the soil profile capable of interacting with N and P dissolved in the percolating water and by the concentrations of dissolved N and P that the soil components maintain in the soil solution. Time of contact between the percolating solution and any soil component will in turn depend on the rates of infiltration and percolation into the soil.

Some of the theories developed to describe water movement in soils can be applied to evaluate the potential loss of N and P from various soil types as a result of subsurface runoff. Gardner (1965) developed equations to describe the movement of nitrate in the soil profile due to leaching. The chemical interactions that occur between dissolved inorganic N and P and soil components (discussed later), when water percolates through the soil, must be taken into consideration. Inclusion of a terms in the equations developed by Gardner (1965) to describe the relationships between N and P in particulate and aqueous phases is therefore necessary. This could take the form of a linear adsorption isotherms relevant to the concentrations of dissolved inorganic N and P maintained in the solution of a particular soil. Biggar and Corey (1969) have also reviewed the literature on infiltration and percolation of water in agricultural soils as it pertains to nutrient movement.

The movement of solid phase material in contact with natural waters occurs during surface runoff and in streams. The amount of solid material capable of entering surface runoff will depend on the intensity of rainfall, physical and chemical attachment between various solid components, and the amounts of energy of runoff waters (Guy, 1970). It is the energy of surface runoff or stream water, however, that governs the amounts of a specific size fraction of particulate materials which will remain in suspension during water flow.

The primary source of particulate material to surface runoff and streams is eroding soil (Guy and Ferguson, 1970), although in urban areas with little ongoing development, particulates may be dominated by specifically urban detrital material (e.g. street litter and dust) and organics derived from urban vegetation.

The total on-site losses of soil due to sheet and rill erosion are not necessarily delivered to streams. The amount of sediment that travels from a point of erosion to another point in the watershed is termed the sediment yield (Johnson and Moldenhauer, 1970). Consequently the Universal Soil Loss Equation used to predict field soil losses on an average annual basis (Wischmeier and Smith, 1965) must be corrected when used to predict sediment loads in streams because deposition of particulates may occur on the land surface as a result of slope variations before surface runoff reaches a stream. It is for this reason that estimates of soil loss in surface runoff from sites within a particular watershed cannot be translated into total N and P losses through a knowledge of the total N and P contents of the soil, if the N and P losses are to be related to P enrichment of surface waters.

An associated complication arises from the fact that soil P is primarily associated with the solid phase. In contrast, NO_3-N is soluble in soil and moves in solution in runoff. As soil erosion is a selective process with respect to particulate size, selectivity has been observed for P loss in surface runoff. The extent of the selectivity depends on the particle sizes with which most of the soil P is associated.

This observation has lead to the concept of enrichment ratios (E.R.), which for P are calculated as the ratio of the concentration of P in the particulate phase of surface runoff to the concentration of P in the source of the particulate phase. This effect was first considered by Rogers (1941) who observed E.R. values of 1.3 for total P and 3.3 for "0.002N H_2SO_4 extractable" P for a silt loam situated on a 20-25% slope. Other values range from 1.5 to 3.1 for total P (Knoblauch et al., 1942; Neal, 1944; Stoltenberg and White, 1953), whereas Massey and Jackson (1952) observed values between 1.9 and 2.2 for "water-soluble plus pH extractable" P for silt loams in Wisconsin. The selective nature of surface runoff with respect to P is due to selective removal of fine soil particulates as a result of the energy limitations of runoff and the fact that a large percentage of total soil P is frequently associated with clay-sized material. A high proportion of the total P in soils may be associated with the clay-sized fraction (Scarseth and Chandler, 1938; Williams and Saunders, 1965; Syers et al., 1969). Greater selectivity of fines and consequently particulate P will occur as the energy of surface runoff decreases. Stoltenberg and White (1953) observed that as precipitation disposed of as surface runoff decreased from 7 cm/hr (2.75 inches/hr) to 0.025 cm/hr (0.01 inches/hr), the clay content of eroded material from a soil with a clay content of 16-18% increased from 25 to 60%. This had obvious implications in relation to the nature of the sediment load carried by a stream and the interactions of P between the solid and aqueous phases, particularly during periods of surface runoff. It should be pointed out, however, that although the P content of the sediment load may increase as surface runoff diminishes, as may be predicted from the work of Stoltenberg and White (1953), the total P load may not change, and could decrease, due to lower sediment loads.

The particulate material carried in streams may be divided into bed load and wash or suspended load. The bed load, which may also have a contribution from existing stream sediment, is that which moves along or close to the stream bed, whereas the wash load is maintained in the flow by turbulence (Johnson and Moldenhauer, 1970). By inference from

the selectivity of surface runoff for fine soil particulates, the wash load will be high during surface runoff events. Furthermore, Johnson and Moldenhauer (1970) suggest that the wash load travels at about the same velocity as the water with which it is in contact. Consequently, P assoicated with the clay- and silt-sized particulates constituting the wash load will move between any two points in the stream profile at the same speed as the ambient dissolved forms of P.

Increased turbulence in streams during high flow, or arising from an increasing gradient, will tend to maintain in suspension particle sizes more characteristic of the bed load, and may even resuspend existing stream bed sediment. In a study of total P loads in the Pigeon River, N.C., Keup (1968) noted that an increase in gradient from 2.81 to 4.35 m/km, over which no tributaries entered the main stream, resulted in a 90.8 kg/day (200 lb/day) increase in the total P load carried.

It appears that in the majority of cases a large proportion of particulate N and P in streams arises from soil erosion. The N and P may be stored in stream bed sediments, but unless the stream is actively aggrading, the amount stored will be less than the inflow (Keup, 1968). That which is stored is liable to resuspension and transport due to turbulence during periods of high flow.

3.5.4 Chemical Factors Affecting the Mobility of P

3.5.4.1 Nature of Soil P

Whenever water containing a particular concentration of dissolved inorganic P comes into contact with soil material, there is a possibility for sorption, desorption, or dissolution reactions to take place. The types of reactions are the same regardless of whether they occur under conditions existing in the soil profile, surface runoff, or streams, all of which may be regarded as soil-water ecosystems. Although in some cases biological assimilation may initially affect the distribution of P between dissolved and particulate phases of soil-water ecosystems, the

distribution of P between these phases will be determined by the nature of the inorganic particulates and the concentrations of dissolved inorganic P in solution (Keup, 1968; McKee et al., 1970; Ryden et al., 1972b).

On the basis of solubility product criteria, it has been postulated that discrete phase crystalline Fe and Al phosphates exist in soils (Hemwall, 1957; Kittrick and Jackson, 1956; Chakravart and Talibudeen, 1962). The general occurrence of discrete Fe and Al phosphates seems doubtful based on the ion product data presented by Bache (1964) and the experimental observations of Hsu (1964). That Fe and Al phosphates are formed as a temporary phase in the vicinity of phosphate fertilizer particles due to conditions of localized high acidity and P concentration is well established (Lindsay and Stephenson, 1959; Huffman, 1969). The compounds, however, will not be stable as the dissolved inorganic P concentration in the soil solution or aqueous portion of other soil-water ecosystems decreases.

The calcium phosphate minerals, apatite (Shipp and Matelski, 1960) and calcic fertilizer-soil reaction products (Huffman, 1969) have been identified in soils. The amounts of apatite are only appreciable in weakly weathered soils (Williams et al., 1969), as predicted by the weathering indices of Jackson (1969). Calcic fertilizer-soil reaction products may be present in neutral and calcareous surface soil horizons, and their importance in maintaining high concentrations of dissolved inorganic P in soil-water ecosystems should not be overlooked.

It has been demonstrated that the uptake or sorption of P from solution by soils is significantly related to the presence of short-range-order (amorphous) oxides and hydrous oxides of Fe and Al (Williams et al., 1958; Gorbunov et al., 1961; Bromfield, 1965; Hsu, 1964; Saunders, 1965). Furthermore, "pure" oxides and hydrous oxides of Fe and Al, and short-range-order aluminosilicates have also been shown to be particularly effective in the sorption of inorganic P from solution (Gastuche et al., 1963; Muljadi et al., 1966; Hingston et al., 1969).

The sorption of inorganic P by Fe and Al oxides and hydrous oxides is known to be rapid, as is the sorption of P by soils. Furthermore, short-range-order Fe and Al oxides and hydrous oxides are ubiquitous in soils (Hsu, 1964), their relative amounts depending on parent material, climatic, and drainage conditions, and occur mainly as coatings on other soil components. Shen and Rich (1962) and Jackson (1963) have noted the occurrence of Al hydroxypolymers, and Dion (1944) and Roth et al. (1969) have reported the presence of Fe oxide and hydrous oxide coatings on clay mineral surfaces. Such coatings, in conjunction with the greater surface area of the clay fraction compared to that of the other particle size fractions 50% of the total P in soils may be assoicated with the clay fraction, as well as the E.R. effect for P as a result of soil erosion. Attempts have been made to correlate P sorption with the clay content of soils (Williams et al., 1958). Correlations between P sorption and clay content after removal of Fe and Al oxides and hydrous oxides often have been poor. Better correlations may be expected if P sorption is related to the content of water-dispersed clay. The sorption of P by water-dispersed clay and silt of soils has obvious implications to reactions occurring between dissolved and particulate P in surface runoff and streams.

Sorption of inorganic P by $CaCO_3$ has also been demonstrated (Cole et al., 1953). The nature of the surfaces of calcite in calcareous soils may be very different from those of pure calcite (Buehrer and Williams, 1936; Lahav and Bolt, 1963; Syers et al., 1972).

Consequently, three basic forms of inorganic P may exist in unfertilized soils (Syers and Walker, 1969; Williams and Walker, 1969), namely apatite, which is a discrete phase P compound; P sorbed on the surfaces of Fe, Al and Ca soil components (non-occluded), and P present within the matrices of Fe and Al components (occluded). In fertilized soils, a variety of P fertilizer-soil reaction products may exist as transient phases. As the solubility product of pure apatite in water is low (0.03 µg P/ml at pH 7, Stumm, 1964) and the P held within the matrices of Fe and Al components virtually chemically immobile, except under reducing conditions in the case of Fe, major emphasis should be

directed towards the reactions involving P in solution and that sorbed on the surfaces of Fe, Al and Ca components as well as the release of P due to dissolution of fertilizer-soil reaction products.

Elucidation of the composition of soil organic P is restricted by lack of extractants capable of removing organic P from soils in a relatively unaltered form and by the inadequacy of current methods for mildly degrading extracted organic P-organic matter complexes. Existing data indicate that most of the organic P in soils is associated, in an ill-defined manner, with the humic and fulvic acid complex of soil organic matter (Anderson, 1967). Of the specific forms of organic P that have been identified in soils, inositol phosphates are present in largest relative amounts, comprising up to 60% of the total organic P (Anderson, 1967; Cosgrove, 1967; McKercher, 1969). Other specific organic P compounds are present in soil in much lower quantities: nucleic acids account for 5 to 10%, and other phosphate esters such as phospholipids, sugar phosphates, and phosphoproteins for less than 1 to 2% (McKercher, 1969).

3.5.4.2 Sorption of Dissolved P by Soils

The sorption of dissolved inorganic P by soils may be described by sorption isotherms. Numerous workers have shown that sorption may be described by some of the adsorption isotherms developed to describe gas adsorption by solids (Russell and Prescott, 1916; Olsen and Watanabe, 1957; Rennie and McKercher, 1959; Syers et al., 1973). Similar observations have been made for the sorption of inorganic P by soil components such as kaolinite and short-range-order Fe and Al oxides and hydrous oxides (Gastuche et al., 1963; Muljadi et al., 1966; Kafkafi et al., 1967). Although these studies have been useful in describing relationships between various soils and soil components with respect to their P sorption capacities, they have provided little information regarding P sorption behavior from solutions containing the low dissolved inorganic P concentrations characteristic of most soil-water ecosystems, largely because of the high levels of added P used (Ryden et al.,

(1972b). Furthermore, Syers et al. (1973) obtained two linear Langmuir relationships which intersected at equilibrium P concentrations varying from 1.5 to 3.2 μg P/ml in the equilibrium solution for three contrasting soils; an observation which probably invalidates interpretations of P sorption made from many previous studies were high levels of added P were used.

The study of White and Beckett (1964), conducted at initial dissolved inorganic P concentrations, comparable to those existing in soil-water ecosystems, provides a useful basis for understanding the interactions between aqueous and particulate phases of P in runoff and streams. White and Beckett (1964) defined the intersection of the P sorption isotherm and the abscissa, the "equilibrium phosphate potential" ($1/2\ pCa + pH_2PO_4$), abbreviated to the "equilibrium P concentration" by Taylor and Kunishi (1971). The intersection is equivalent to the inorganic P concentration in the ambient aqueous phase when there is no net sorption or release of P, i.e., $\Delta P=0$. This is a point of reference which provides a predictive estimation of sorption or release of P, should the P concentration in solution change. Furthermore, the average slope of the sorption curve over a given final P concentration range provides information on the ability of the soil to maintain the P concentration of the equilibrium P concentration. The steeper the slope, the closer will the final P concentration be to the equilibrium P concentration. The slope of the curve, although not related to total P sorbed, is related to the extent to which that soil may sorb P over the concentration range considered. The potential of this approach in predicting the chemical mobility of P in soil-water ecosystems is clearly evident and has been used with regard to streams by Taylor and Kunishi (1971) and Ryden et al. (1972a, b) for rural and urban soils, respectively.

The desorption of sorbed P is not as simple as may be inferred from the sorption-release relationships obtained by White and Beckett (1964). In fact very few studies have been reported regarding the desorption of sorbed P. Syers et al. (1970) observed that, for a range

of soils with widely differing P sorption properties, those which sorbed the largest amounts of P desorbed the smallest percentages of that sorbed during a subsequent desorption step. A similar relationship was noted by Ryden et al. (1972a). These studies, however, involved a desorption step following sorption of P from solutions containing P concentration in excess of those commonly found in soil-water ecosystems.

In studies involving the sorption of P by kaolinite from solutions containing realistic inorganic P concentrations. Kafkafi et al. (1967) observed that initially all the sorbed P was isotopically exchangeable. During a subsequent washing or desorption step, however, a portion of the sorbed P became nonexchangeable, or "fixed", this portion being dependent upon the amount of P sorbed, the number of washings and the nature of the previous P sorption cycle. P sorption was represented by either one-step P sorption from a range of solutions of different initial P concentration or by successive additions of small amounts of dissolved inorganic P. Both these forms of P sorption, as well as an effect analogous to washing, could occur in soil-water ecosystems.

Although the mechanisms involved in the retention of organic P by soils have not been established fully, there is evidence that inositol hexaphosphate, and possibly other organic P compounds, are retained by a precipitation rather than by a sorption reaction. Nevertheless, removal of dissolved organic P from solution appears to be a rapid process. Pinck et al. (1941) reported that many commonly occurring water-soluble organic phosphates, e.g. salts of glycerophosphate, hexose diphosphate, and nucleic acids, become non-extractable with water at almost the same rate and as completely as dissolved inorganic P. The retention of water-soluble organic P by sorption reactions may occur by at least two basically different mechanisms (Sommers et al., 1972). Goring and Bartholomew (1950) observed that removal of "free iron oxides" considerably reduced the amount of fructose-1, 6-diphosphate sorbed by subsoil material, suggesting that the sorption of organic P may occur through orthophosphate groups by a similar mechanism to that for inorganic P.

It is possible that organic P can be retained by interaction of the organic moiety of the phosphate ester with inorganic soil components. For example, nucleic acids and nucleotides are protonated at a pH 5 (Jordan, 1955) and could consequently be retained on clay surfaces by displacement of exchangeable cations. Furthermore, physical sorption, also through the organic portion of the molecule, is possible, particularly if the molecular weight of the compound is high, as suggested by Greenland (1965). In such cases retention is weak and is accomplished by van der Waals and ion-dipole forces. Greaves and Wilson (1969) have implicated physical adsorption in the retention of nucleic acids by montmorillonite. It is also possible that retention occurs indirectly through other soil organic compounds such as fulvic and humic acids after interaction of organic phosphates with these species (Martin, 1964).

The desorption of sorbed organic P has not been extensively studied. The hypothesis that inorganic P added to soils displaces sorbed organic P to solution (Latterell et al., 1971) was not supported by the data presented by Wier and Black (1968). Although organic P may be leached from soils, it appears that a large proportion of that removed may not be in a dissolved form. After incubating sucrose with ammonium nitrate in the upper portion of a calcareous soil, Hanapel et al. (1964) found that most of the organic P removed by leaching was present in a particulate rather than a dissolved form.

3.5.4.3 Chemical Aspects of P in Subsurface and Ground-Water Runoff

Losses of P in subsurface and ground-water runoff have been considered minimal in the past, but as discussed in section 3.4 such losses can amount to a significant proportion of losses from agricultural land, and possibly a major proportion from forest lands. The supposition that P losses in subsurface and ground-water runoff are low probably stems from the concept of P immobility based on the P sorption properties of soils using added inorganic P concentrations far in excess of those normally present in the soil solution.

It is of interest to note that many of the reported mean concentrations of dissolved inorganic P in subsurface runoff are within the range of values expected to be maintained in the soil solution. Pierre and Parker (1927) reported values ranging from 0.020 to 0.350 µg P/ml, with an average of 0.090 µg/ml, for several surface soils from the southern and midwestern states of the USA. These workers also noted that dissolved inorganic P concentrations could be maintained at a fairly constant level. Barber et al. (1962) reported similar values for the upper 15 cm (6 inches) of 87 soils from the midwestern USA, with an average of 0.180 µg P/ml; the frequency distribution of the values obtained, however, suggested a mode of between 0.040 and 0.060 µg P/ml.

As water percolates through the soil profile, there tends to be a "chemical sieving" of dissolved inorganic P (Black, 1970). This arises as a result of the sorption of inorganic P by soil components. The low concentrations of P found in ground-water runoff, which has experienced the maximum effects of deep percolation with concomitant increase of contact with P-deficient particulates of the subsoil, are undoubtedly a direct result of the chemical sieving effect. This is illustrated by other data presented by Barber et al. (1962). For the same 87 soils mentioned previously, the average dissolved inorganic P concentration at a depth of 46-61 cm (18 - 24 inches) was 0.089 µg/ml, less than half that for the upper 0-15 cm (0 - 6 inches). A similar effect is observed in results presented by Ryden et al. (1972a) for the P sorption properties of successive soil horizons of Miami silt loam.

The concentration of dissolved inorganic P in subsurface and ground-water runoff will depend on the nature and amounts of P-retaining components in the profile, the surface area exposed to percolating waters, and the ease of percolation which affects the contact time of dissolved inorganic P with the retaining components. In studies of P leaching through columns of organic soils, in the laboratory, Larsen et al. (1958) observed that P retention, measured by ^{32}P autoradiographs, was closely correlated with the total hydrous Fe and Al oxide ("sesquioxide") content. Similarly. losses of P due to leaching through a deep silicious sandy soil were

demonstrated in W. Australia by Ozanne (1962). When 225 kg/ha (209 lb/ac) of ^{32}P-labelled superphosphate was broadcast during winter on a fallow sandy soil, over 50% of the P had penetrated to more than 1 m (3 ft) below the surface within 38 days, during which 230 mm (9 inches) rain had fallen. Ozanne (1962) also demonstrated that the potentially large losses of P to subsurface and ground-water runoff from sandy soil compared to that from loamy soils were due to quantitative rather than qualitative differences in P-retaining components.

Although major emphasis has been placed on P losses in surface runoff, it appears that losses of P to subsurface and ground-water runoff, although of little significance from an agricultural standpoint, may under certain conditions constitute a significant loss of P from agricultural watersheds in terms of the P enrichment of surface waters. Losses of P to subsurface and ground-water runoff are even more difficult to evaluate than those in surface runoff and demand further investigative attention.

3.5.4.4 Chemical Aspects of P in Streams and Stream Bed Sediments

As discussed previously, surface runoff from agricultural land constitutes a heterogeneous and relatively short-lived system. Any attempt to consider the distribution and chemical mobility of P between solid and aqueous phases before entry into the receiving stream would be pointless as a new and more homogeneous system is rapidly established. Surface runoff in urban areas is somewhat different because in most cases it is channellized shortly after origin by alteration of surface drainage patterns, and under such circumstances it is analogous to a stream in an artificial channel. Consequently, the chemical mobility of P will be discussed from the standpoint of the stream environment.

The potential of suspended particulates derived from eroding soil to modify the dissolved inorganic P concentration of streams has been suggested by Taylor (1967) and Biggar and Corey (1969). Wang and Brabec (1969) also implied that inorganic P was sorbed by suspended

particulate material from observations of dissolved inorganic P concentrations in the Illinois River at Peoria Lake.

An evaluation of the possible effects of eroded soil materials on the dissolved inorganic P concentrations of streams may be obtained from P sorption studies (Taylor and Kunishi, 1971; Ryden *et al.*, 1972a, b). It is essential, however, that conditions realistic of those existing in streams are used if meaningful results are to be obtained (Ryden *et al.*, 1972a). Widely differing interpretations can be made as solution:soil ratios and initial P concentrations are changed from those conventionally used in P sorption studies to those realistic in terms of the stream environment. In an investigation of P sorption in a Miami silt loam soil by Ryden *et al.* (1972), the data suggested that inorganic P released from the A1 horizon, which contained a P fertilizer-soil reaction product, would be largely resorbed by the noncalcareous B1 horizon and to some extent by the calcareous 3C1 horizon, should the horizons erode together. Sorption studies employing low initial added inorganic P concentrations and a wide (400:1) solution:soil ratio indicated that the B1 horizon has a much lower ability to remove dissolved inorganic P from solution than expected, this being equal to or only slightly greater than that of the 3C1 horizons. It was found (Ryden *et al.*, 1972b) that the A1 and 3C1 mixtures were able to maintain lower dissolved inorganic P concentrations than the A1 and B1 mixtures. The conditions used by Ryden *et al.* (1972a, b) to predict the potential of eroding soils to modify the dissolved inorganic P concentrations of streams gave results which were comparable to those obtained in simulated stream systems using a solution:soil ratio of 1000:1. This is equivalent to a sediment concentration of 1000 mg/l, which lies well within the range of values cited by Guy and Ferguson (1970) and Johnson and Moldenhauer (1970).

The P sorption studies reported by Taylor and Kunishi (1971) and Ryden *et al.* (1972a, b) involved closed systems, i.e., soil in contact with the same aqueous phase. This may be justified on the grounds that the wash load of a stream travels at the same velocity as the water in which it is suspended (Johnson and Moldenhauer, 1970) as discussed previously.

Sorption studies may be used to provide reasonable estimates of dissolved inorganic P concentrations in streams, under various flow conditions, draining rural watersheds. Taylor and Kunishi (1971) observed that dissolved inorganic P concentrations, during base flow of a stream draining a small agricultural watershed in Pennsylvania, were in the range of 0.040 to 0.060 µg P/ml, values which were close to those predicted from P sorption studies using stream bank sediment and subsoil material. During periods of surface runoff, predicted dissolved inorganic P concentrations would be in excess of 0.20 µg P/ml for the surface soil used by Taylor and Kunishi (1971) and 0.10 µg P/ml for that used by Ryden et al. (1972a) due to release of P from eroded surface soil; however, predictions from the work of Taylor and Kunishi (1971) are based on the use of a narrow (10:1) solution:soil ratio. The ability of eroding stream bank material or resuspended stream bed sediment to resorb inorganic P released to solution should not be ignored (Taylor and Kunishi, 1971).

It is important to distinguish between the quantities of various types of soil materials expected to enter streams in urban as opposed to agricultural surface runoff. In agricultural areas, surface runoff will carry primarily surface soils to receiving streams. Surface soils may frequently contain fertilizer-soil reaction products capable of producing significant increases in dissolved inorganic P concentrations, due to their dissolution (Ryden et al., 1972a). In urban areas, however, land under development, which is prone to severe erosion, is frequently graded, exposing some or all horizons of the area profile to potential erosion. Dissolved inorganic P concentrations of receiving streams in urban areas may be sufficiently high that the addition of eroded soil material may cause a reduction in the dissolved inorganic P concentration.

An approach similar to that used by Taylor and Kunishi (1971) and Ryden et al. (1972a, b) could be used to identify other diffuse sources of potential P enrichment within a watershed. The approach could be particularly useful for estimating the potential of various forms of urban detrital material to influence the dissolved inorganic P concentrations of surface runoff.

One diffuse source of considerable importance is the leachate from leaves, particularly during the autumn. An appreciable percentage of the total P in leaf tissue may be in a water-soluble form. Ash leaves may contain 61.5% of total P as water-soluble inorgaic P (Nykvist, 1959). Cowen and Lee (1972) observed that 44 and 120 µg soluble inorganic P per g air-dry weight of fallen oak and poplar leaves, respectively, could be leached by 1 liter of distilled water percolating at a rate of 8.4 ml/min. Greater amounts of P were released from oak leaves during consecutive leaching cycles, and after fragmentation of whole leaves. Similar experiments were conducted by Timmons et al. (1970) using agricultural crop residues. These were leached in a fresh condition and after dyring, and freezing and thawing cycles. The data suggest that the leaching of crop residues is most likely to contribute to the dissolved inorganic P concentration of streams during spring thaw when, after numerous freezing and thawing cycles, the residues will be carried over frozen ground in surface runoff. When greater infiltration can occur, a portion of the leached P may be retained in the soil due to sorption.

Little is known of the chemistry of stream-bed sediment although it is conceivable that it is similar to that of the subsoil of the surrounding area (Taylor and Kunishi, 1971). Consequently, P sorption studies using subsoil material may provide some information on the role of stream-bed sediment in regulating the dissolved inorganic P concentration due to its suspension during turbulence. This would be particularly true in watersheds, with little contribution to stream-bed sediment as a result of surface runoff. In watersheds where surface runoff is a regular occurrence, however, stream-bed sediment is expected to have a significant contribution from surface horizon soil material, and the latter could contribute to base flow concentration of inorganic P.

Care should be taken, however, in the extension of the P sorption properties of field soils to stream-bed sediment. Hsu (1964) observed that the amount of inorganic P sorbed by soil after storage for 1 year in a continuously wet condition, increased from 69 to 99 µg P/g soil.

The increased sorption was attributed to release of Fe to solution from crystalline phases due to the development of localized reducing conditions during storage, and reprecipitation of "ferric hydroxide" on contact with more aerobic conditions. The redox status of stream-bed sediments does not appear to have been studied, but it is reasonable to suggest that reduction occurs at depth in the sediment with the possibility of crystalline ferric components being transformed to short-range-order ferrous forms. The importance of short-range-order oxides and hydrous oxides of Fe in the sorption of inorganic P has already been discussed. The possible transformation of Fe from crystalline to short-range-order forms represents the first stage of the more aggressive transformations which occur in lake sediments under anaerobic conditions (Shukla et al., 1971).

The observation of Kafkafi et al. (1967) that the washing of kaolinite, on which P had been sorbed, produces a "pool" of nonexchangeable P is also of direct relevance to the dynamics of P in stream-bed sediments, assuming a similar effect occurs. Stream-bed sediment with associated sorbed P could undergo a series of steps equivalent to sequential washing due to resuspension and settling as a result of minor turbulence. The observations of Kafkafi et al. (1967) suggest that sorbed P could become progressively less exchangeable and may constitute an essentially permanent removal of dissolved inorganic P from streams.

When stream-bed sediment contains eroded fertilized soil materials, however, a different situation may prevail. Ryden et al. (1972a) showed that release of P from a surface soil horizon by repeated washing with P-free 0.1M NaCl initially followed first order kinetics, suggesting that release was due to the dissolution of solid phase P, probably a fertilizer-soil reaction product.

3.5.4.5 Forms of P in Runoff and Streams

In many studies concerned with various aspects of P in runoff and streams there has been a tendency to measure total P. The measurement

of total P discharged by streams does not provide any indication of the amounts of P available for aquatic plant growth. Consequently, the forms of P measured in streams that enter a lake or reservoir are of direct importance in assessing the impact of runoff-and stream-derived P on a body of standing water. Dissolved inorganic P is one of the obvious choices because this form of P is directly available for biological utilization. Objections to the measurement of dissolved inorganic P, as is conventionally determined, have been raised by Frink (1971) on the basis that distinction between dissolved and particulate forms is based on filtration through a 0.45 µm filter. Although it is possible that filtration does not strictly differentiate between dissolved and particulate P, it provides a more realistic measure in terms of the effects of runoff-and stream-derived P on the biological productivity of standing waters than the measurement of total P.

Vollenweider (1968) has also pointed out the necessity of distinguishing between total P and dissolved forms of P because it is possible that P exports from some watersheds occur mainly in biologically unavailable forms, such as apatite. He notes that P exports from the Alpine portion of the Rhine Basin amount to 1.45 kg/ha/yr (1.29 lbs/ac/yr). As this is mainly in the form of apatite, however, the contribution of biologically available P to Lake Constance is relatively small. In other regions it appears that a high proportion of particulate inorganic P in streams draining urban and rural watersheds may in fact be apatite. Eroding urban soils in the Lake Mendota watershed, Wisconsin, contain between 6 and 80% of the total inorganic P as apatite, with amounts exceeding 50% in the lower B and C horizons (Ryden and Syers, unpublished data). For the same soil materials, Sagher and Harris (1972) observed that algal cultures suffered P starvation when the sole P source in the growth medium was C horizon material, indicating the very low biological availability of the P present in apatite.

Chemical fractionation schemes have been used to determine the forms of inorganic P in soils. These schemes evolved from the observations of Chang and Jackson (1957) that certain chemical reagents were able to

solubilize inorganic P contained in various synthetic phosphates and phosphate minerals. Recent workers (Bromfield, 1967; Williams et al., 1971a, b; Syers et al., 1972) have questioned the validity of the separation of inorganic P into Al-, Fe-, the Ca-bound forms, as proposed by Chang and Jackson (1957). Providing that the problems inherent in inorganic P fractionation schemes are recognized, useful interpretations may be drawn from the data obtained. The form of particulate inorganic P which is expected to have the greatest potential impact on the biological productivity of standing waters is that which is non-occluded. Part of the non-occluded inorganic P associated with ferric components is released into solution when anaerobic conditions develop subsequent to sedimentation. Appropriate inorganic P fractionation schemes applied to suspended stream sediments may provide a more meaningful measure of the forms and amounts of particulate inorganic P carried in streams. As pointed out by Taylor et al. (1971), suspended sediment concentrations are frequently not high enough to provide adequate amounts of material in a manageable volume of water. Evaluation of the forms of P in soil materials which are known to be transported to streams in surface runoff may overcome this problem to some extent. In the case of eroding soil materials, the inorganic P fractionation schemes should not be applied to the whole soil, due to the E.R. effect resulting from erosion. Water-dispersed particle size separates should be used.

In spite of the possible errors involved in a dissolved-particulate P split based on filter pore size, it seems that in the majority of cases the most meaningful and useful measurements of P in runoff are dissolved forms, particularly dissolved inorganic P. Frequently dissolved forms of P account for a major percentage of total P (Sylvester, 1961; Sullivan and Hullinger, 1969), whereas dissolved inorganic P in many cases constitutes a major proportion of the total dissolved P. It should be noted that dramatic changes can occur in the concentration of dissolved inorganic P and other P fractions after sample collection, even after only a short period of time. In some cases when samples are not analyzed immediately after collection, the only valid P parameter that can be measured is total P.

3.5.5 Chemical Factors Affecting the Mobility of N

3.5.5.1 Nature of Soil N

Most of the N in soils (perhaps more than 95% of the total N) is bound in organic forms (Biggar and Corey, 1969; Bartholomew and Clark; 1956; Keeney, 1970). The organic N consists of the biologically important nitrogenous compounds (e.g. proteins) and N contained in humic matter. Amino acid N (including proteins) accounts for 20 to 50% of the organic N. About 4 to 10% of the organic N occurs in amino sugars. A wide range of other naturally occurring compounds have been found, but these compounds account for only about 1 to 2% of the total organic N. Consequently, about one-half of the organic N can not be accounted for in known biological compounds. Apparently, during chemical and microbial transformations of soil organic matter, microbially synthesized products are converted to more stable humic forms, including humic and fulvic acids.

The main forms of inorganic N in soils are NH_4^+ and NO_3^-, both of which are available to plants. Small amounts of NO_2^- are sometimes present. The NO_3^- ion shows little tendency for sorption onto soil particles while NH_4^+ is held as an exchangeable ion on soil cation exchange sites. Soils also retain some NH_4^+ within clay lattices. This clay-fixed NH_4^+ is relatively unavailable to plants and microorganisms. The amounts of NH_4-N and NO_3-N present in a soil at a given time generally represent less than 0.1% of the N in the soil. However, organic N is converted slowly to NH_4-N and subsequently to NO_3-N through microbial processes.

3.5.5.2 Transformations and Mobility of Soil N[1]

The predominant storage reservoir of N in soils is the organic matter. Added plant and animal debris are attacked readily by heterotrophic microorganisms with some of the N being added to the organic matter

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[1] Based on the review of Keeney (1970).

reserve, and some being liberated as inorganic (predominantly NH_4^+) waste products. Depending on the supply of available energy (carbon), inorganic N can be immobilized in microbial tissue which continually adds to the supply of organic matter. Concurrently, soil organic matter is continually depleted through microbial breakdown. The dynamic nature of nitrogen turnover in soils does not at first appear to be consistent with the relative stability of soil organic nitrogen. In soils, only about 1 to 3% of the organic N is available for plant uptake in a growing season (Bremner, 1967). The stability of organic N in soils is generally attributed to formation of herterocyclic N compounds, clay-organic matter complexes which are resistant to microbial attack, and to lack of sufficient carbonaceous energy material for complete breakdown (Bremner, 1967).

The processes leading to formation of NO_3^- by microbial action in soils and waters can be grouped conveniently into ammonification and nitrificiation reactions (Delwiche, 1970). Ammonification (organic N to NH_4^+) is conducted by numerous heterotrophs under a wide diversity of environmental (pH, temperature, moisture) conditions (Bartholomew, 1965; Campbell and Lees, 1967). With few exceptions, nitrification (NH_4^+ to NO_2^- to NO_3^-) is carried out by strict chemoautotrophs. Although several heterotrophic nitrifiers have been identified, their importance in the nitrification process is believed to be minimal except in organic-rich environments such as composts and manure piles. Because of its agronomic importance, the nitrification process has been studied widely and reviewed extensively (Quastel and Scholefield, 1951; Campbell and Lees, 1967; Keeney and Gardner, 1970). In contrast to ammonification, nitrification is easily inhibited by unfavorable conditions. Optimal nitrification occurs at pH values near 7 and in the presence of adequate oxygen. The nitrification rate increases with temperature to 35 C, but is inhibited completely at greater than 40 C. Formation of NO_2^- from NH_4^+ (Nitrosomonas spp.) can be inhibited by numerous organic and inorganic compounds. Oxidation of NO_2^- to NO_3^- (Nitrobacter spp.) can be inhibited by the high pH-free NH_4^+ conditions occurring in the

area around urea or anhydrous ammonia fertilizer bands giving rise to phytotoxic NO_2^- accumulation in some alkaline soils following N fertilization.

The salts of NO_3^- are water soluble and nitrate does not react to any great extent chemically or physically with soil constituents to form insoluble products. As a result, NO_3^- in soils is readily available for plant uptake or loss to ground waters by leaching. It is this unique property of NO_3^-, coupled with its toxicological and nutrient properties, that make it of such concern.

The process of denitrification (reduction of NO_3^- or NO_2^- to elemental N and N oxides) provides the major pathway whereby N is cycled back into the atmosphere. Denitrification can occur by two distinct pathways. Biological denitrification occurs when oxygen tensions reach very low levels (through flooding of soils and utilization of oxygen by decomposition of organic matter) and the heterotrophic microorganisms must utilize other electron acceptors. With few exceptions, carbonaceous material must be present to provide sufficient energy (Broadbent and Clark, 1965). In acid soils and waters, NO_2^- is extremely unstable and decomposes to yield NO_3^- and gaseous N oxides, or reacts with soil constituents with the fixation of N in organic matter and formation of N oxides (Bremner and Nelson, 1968; Brodbent and Clark, 1965). These latter reactions have been termed "chemo denitrification" as they do not involve microbial transformations directly and can occur in arable soils. The importance of denitrification reactions in soils and waters cannot be overstated, and these complex and poorly understood reactions must be evaluated to enable recommendations to be made on methods of biological NO_3^- removal from soils and waters as well as N fertilizer rates.

A point worthy of emphasis in the N cycle is that once N enters the soils from any source, its identity is lost because of the complex series of transformations it may undergo. Thus, investigations which attempt to pinpoint sources of NO_3^- polluting water supplies are particularly difficult and, unless properly conducted, subject to many errors (Keeney, 1970).

3.5.5.3 N in Surface Runoff, Streams, and Stream Bed Sediments

Surface runoff occurs whenever the rate of precipitation exceeds the rate of infiltration (Keeney and Walsh, 1972). Soil texture, slope, moisture content, physical character of the surface, soil and water conservation practices, and vegetative cover affect the infiltration rate. Infiltration is usually quite rapid with coarse-textured sandy soils and with well-aggregated, fine-textured soils having large pores and no impervious subsoil layers. The energy associated with the impact of falling raindrops is an important factor in the breakdown of soil aggregates, increasing runoff. Any cover, vegetative or mulch, that absorbs raindrop impact tends to protect the soil structure at the surface and thus promote infiltration.

The high water solubility of NO_3^- and NH_4^+ tends to favor movement of these ions into the soil before runoff occurs (Bailey, 1968; Armstrong and Rohlich, 1970; Biggar and Corey, 1969). Under some conditions, the concentrations of NO_3-N and NH_4-N in runoff may be lower than that in rain water. However, Timmons et al. (1970) noted that leaching of soluble nutrients from cover crops could contribute substantial concentrations of N and P to surface runoff. Furthermore, data on N concentrations in streams receiving runoff from agricultural areas shows that appreciable amounts of N may be transported in surface runoff (See Section III-4). For example, Jaworski et al. (1969) observed NO_3-N concentrations as high as 3 mg/l and frequently near 1 mg/l for Catoctin Creek, Md., a stream draining a predominantly agricultural area. In basins containing large areas of agricultural lands. The nutrient contributions to the receiving water may be appreciable even though the amount per unit of land area (kg/ha) is small.

Dissolved and particulate N compounds in runoff are potential sources of N to streams and lakes. Due to rapid nitrification in aerobic systems, concentrations of NH_4-N in streams are generally low compared to NO_3-N. Amounts of dissolved and particulate organic N are variable. Total N (including particulate) is frequently considerably higher than NO_3-N.

The N that reaches surface waters may be immobilized in the sediments and thus only partially available (Keeney, 1972). Conversly, some of the N may be released from the sediments by microbial mineralization. Soil erosion is a selective process, and eroded material contains 2.7 to 5 times the total N content of the original soil (Hensler and Attoe, 1970).

The effect of fertilizer N for continuous corn on runoff and nitrogen losses has been studied in Missouri. It was found that 0.5 m^3/ha (7 ft^3/ac) of runoff resulted from 11.4 cm (4.5 inches) of rain and that a runoff loss of only 0.01 kg/ha (0.01 lb/ac) of NO_3-N resulted from application of 198 kg/ha (177 lb/ac) of fertilizer N (Smith, 1967).

Minnesota workers found soil losses of 7.7 metric tons/ha (7 tons/ac) for fallow conditions, 2.0 (1.8) for continuous corn, and less than 0.55 (0.5) for crop rotations. The N in this sediment amounts to 63.5 (56.7), 12.9 (11.5) and less than 5.6 kg/ha (5 lb/ac), respectively. The dissolved N in the runoff water ranged from 0.75 to 1.21 kg/ha (0.67 to 1.08 lb/ac) for the continuous corn and for corn and oats in the rotation. However, the amount of dissolved N in the runoff water from hay was 3.47 kg/ha (3.10 lb/ac). This was probably due to the leaching of N from the frozen hay residues (Timmons et al., 1970). Wisconsin researchers have also noted that the content of soluble N in runoff water from sod plots was about 5 times greater than that from fallow plots, while work on rotational land with slopes ranging from 3 to 20% has shown that runoff losses of NO_3-N were less than 1 kg/ha (0.9 lb/ac) (Hensler and Attoe, 1970).

3.5.5.4 Nitrogen in Subsurface and Ground-Water Runoff

Water that infiltrates into the soil ultimately is either evaporated, transpired, seeps along impervious layers, or percolates to the water table. The amount of leaching at a particular location is largely related to conditions affecting evapotranspiration, soil physical conditions, and to precipitation distribution and intensity. As reviewed by Keeney (1970), the negatively charged nitrate ion (NO_3^-)

is soluble in water nad generally moves with the water. The positively charged ammonium ion (NH_4^+), however, is held in exchangeable form by clay minerals and soil organic matter which greatly restrict its movement. Sediments and organic matter do not readily move downward in the soil; bacteria and pesticides are generally adsorbed by soil particles; and the bulk of most plant nutrients are held in exchangeable, non-exchangeable, adsorbed or organic forms that are resistant to leaching. Thus, NO_3-N is usually of greater concern in groundwater pollution.

Nitrate-N tends to accumulate at the soil surface during prolonged droughts due to evapoaration. Rain following a drought moves NO_3-N into the root zone, or beyond in the case of excessive rains. Incorporation of carbonaceous crop residues tends to retard production of NO_3-N and movement due to microbial immobilization of N.

In Colorado, the average kg/ha (lb/ac) of NO_3-N found in the soil to a depth of 6.1 m (20 ft) for various cropping practices were: alfalfa, 88 (79); native grassland, 100 (89); cultivated dry land, 292 (261); irrigated fields not in alfalfa, 569 (508); and corrals, 1,608 (1,436) (Stewart et al., 1968). Variations in the NO_3-N content of the corral soils were directly related to teh degree of aeration. For the irrigated fields the annual losses of NO_3-N to the groundwater were 28 to 41 kg/ha (25 to 37 lb/ac). Stout and Burau (1967) found that soil fertility levels are the prime factors determing the amount of NO_3 below the root zone, showing that NO_3 is largely from natural sources.

In preliminary study in Wisconsin (Olsen, 1969), undisturbed, cultivated, and poorly drained and well drained barnyard soil profiles were analyzed for their NO_3-N content. The respective profiles averaged 55 (49), 231 (206), 165 (147), and 134 kg/ha (120 lb/ac) of NO_3-N to a depth of 2.4 m (8 ft). The content tended to decrease with soil depth, apparently due to plant uptake, denitrification, and microbial immobilization processes.

On irrigated sands, loss of NO_3-N is closely related to the amount of fertilizer N and irrigation water applied (Olsen, 1969).

In Wisconsin 30.5 cm (12 inches) of rain or irrigation water during a 5-week period following application of ammonium nitrate to a fallow Plainfield sand caused most of the NO_3-N to move down to the 1 to 1.5 m (40 to 60 inch) depth. An additional 15 cm (6 inches) of precipitation during the following month moved most of the NO_3-N below the 1.8 m (6 ft) depth. In a related study, the annual application of 336 kg/ha (300 lb/acre) of N as ammonium nitrate to a silt loam soil for corn resulted in the accumulation of excessive amounts of NO_3-N in the subsoil, compared to very little accumulation from the 112 kg/ha (100 lb/acre) rate. The average annual precipitation was about 81.3 cm (32 inches) and the rate of movement of NO_3-N was about 41 cm (16 inches) annually. These data suggest that pollution of the groundwater with NO_3-N can be greatly reduced by limiting the rate of fertilizer N to approximate the needs of the crop.

Hedlin (1971) found that high NO_3-N concentrations in groundwater were most likely where repeated heavy applications of organic matter or nitrogenous materials take place (e.g. in the vicinity of farmsteads, towns or villages). He also noted that the NO_3-N concentration in the groundwater decreases rapidly as one moves away from the source, indicating either dilution by NO_3-free water or denitrification or both. Little is known of the persistence of NO_3 in groundwater and evaluation of the possibility of denitrification in groundwaters would seem to be a needed area of investigation.

3.6 Natural Renovation Mechanisms Available to Remove Nutrients From Agricultural Runoff

The physical and chemical processes controlling the mobility of soil phosphorus and nitrogen are reviewed in Section 3.5. Included in this section is a discussion of natural mechanisms which tend to limit the movement of P and N in agricultural runoff. These processes include the soil stabilization and sediment deposition, sorption of phosphate by soil particles, retention of NH_4-N by soil cation exchange sites, immobilization of nutrients by plant assimilation, and denitrification of NO_3-N.

Soil erosion is a major factor in the transport of nutrients in agricultural runoff. Thus, mechanisms and practices which minimize erosion and transport of soil particles reduce nutrient transport in runoff. The natural processes of major importance is maintenance of a crop or mulch cover on the soil (Amemiya, 1970).

Phosphate characteristically exhibits a low water solubility in soil systems as a result of sorption and/or precipitation processes (Taylor, 1967). However, the retention of phosphate is a function of soil type and is generally high for fine-textured soils and low for sands. Thus, the success in using sorption by the soil as a mechanism for removing P from runoff will depend on the characteristics of the soil. In well drained silt loam soils, sorption should reduce P concentrations to sufficiently low levels in the absence of excessive loadings of P to the soil.

Nitrification greatly increases the mobility of NH_4-N and organic N in soils and the tendency for N to percolate to ground waters. Conditions for nitrification are generally favorable for agricultural crops. Mobility is best controlled by balancing NO_3-N from the soil but requires anaerobic conditions which are generally unacceptable for crop production.

Practices based on the factors and mechanisms controlling the mobility of nutrients in soils and used to minimize nutrient transport in runoff are discussed in Section 5.

3.7 Probable Changes in Land Use Activities (1980, 2000, 2020)

Changes in the total cropping area for the U.S. region of the Great Lakes Basin were estimated in the Great Lakes Basin Framework Study, Appendix 13, Land Use and Management (1971) and are shown in Table 11. A gradual decrease in cropping area was predicted, amounting to 10 per cent of the present cropping area by 2020. Some changes in areas for crop groups were predicted. Areas in idle cropland, pastures and meadows, and close grown crops were projected to decrease, while row crop area was expected to increase. These projections indicate that there is little land available for conversion to agricultural use in the Basin.

Table 11. Total Cropping Area, Current and Projections for 1980, 2000 and 2020 for Great Lakes Basin[1]

Crops	Total Area							
	1966		1980		2000		2020	
	1000 hectares				(1000 acres)			
Row Crops	3,949	(9,750)	3,849	(9,503)	4,084	(10,083)	4,821	(11,903)
Close Grown Crops	1,458	(3,600)	1,617	(3,993)	1,467	(3,621)	1,284	(3,170)
Pastures and Meadows	2,719	(6,715)	1,995	(4,926)	1,721	(4,239)	1,654	(4,083)
Orchards and Vineyards	243	(600)	161	(398)	165	(407)	185	(456)
Idle Cropland	3,219	(7,947)	3,620	(8,939)	3,319	(8,194)	2,398	(5,920)
Total	11,588	(28,612)	11,242	(27,759)	10,750	(26,544)	10,342	(25,532)

[1]Great Lakes Basin Framework Study, Appendix 13, 1971.

3.8 Projected Seriousness of Future Pollution Problems Derived From Nutrients From Agricultural Lands.

The amounts of nutrients from agricultural lands expected to reach the Great Lakes can be projected from land use predictions (Section 3.7) and estimated loadings per unit area of agricultural land (Section 3.4). These estimates assume that the rate of loading for a given type of crop will not change. Further, as discussed in Section 3.4, the estimated loadings are only approximations and not sufficiently accurate to justify quantitative estimates of changes in loadings due to changes in fertilizer use or other farming practices. Because land use changes are expected to be small, the corresponding changes in nutrient loadings from agricultural lands are also small (Table 12). On this basis, it appears that the nutrient loadings from agricultural land have reached a maximum value and will remain the same, given current agricultural practices. Relatedly, if agricultural technology can be developed to reduce nutrient loadings to surface waters per unit land area, corresponding decreases in nutrient loadings to the Great Lakes can be expected.

While the nutrient loadings from agricultural lands may remain relatively constant, the proportion of phosphorus entering the Great Lakes from agricultural lands as compared to other sources will likely increase substantially. Treatment of municipal and industrial wastes to remove phosphorus is receiving major emphasis. These sources presently account for a high proportion of the phosphorus entering the Great Lakes. Phosphorus budgets for the total Great Lakes system (Great Lakes Water Quality Board, 1973) do not provide estimates of the proportion of the total P loading derived from municipal wastes. Further, discrepancies exist between the budgets presented in the section on water quality and in the section on eutrophication, which includes estimates in reduction in P loading for the period 1971-1973. However, it appears that an overall reduction of about 25% in the P loading to the Great Lakes from the U.S. and Canada was projected by 1973. On this basis, if agriculture previously contributed about 20%, this proportion increased to about 25% if projected reduction was achieved. From another viewpoint, at least an 80 percent

Table 12. Projected Future Loadings of Soluble Nutrients to the Great Lakes from Agricultural Lands.

Crops	Estimated Nutrient Loadings to the Great Lakes							
	1966[1]	1980[2]	2000[2]	2020[2]	1966[1]	1980[2]	2000[2]	2020[2]
	metric tons/yr (tons/yr)							
	Inorganic Phosphorus				Inorganic Nitrogen			
Row Crops	829 (914)	808 (889)	858 (944)	1,010 (1,111)	6,160 (6,780)	6,000 (6,600)	6,370 (7,010)	7,520 (8,270)
Close-Grown Crops	190 (209)	210 (231)	191 (210)	167 (184)	2,400 (2,680)	2,700 (2,970)	2,450 (2,700)	2,140 (2,350)
Pastures and Meadows	598 (659)	439 (483)	379 (419)	364 (400)	6,250 (6,880)	4,590 (5,050)	3,960 (4,360)	3,800 (4,180)
Orchards and Vineyards	51 (56)	34 (37)	35 (39)	39 (43)	379 (417)	251 (276)	257 (283)	289 (318)
Idle Crop Land	161 (178)	181 (199)	166 (183)	120 (132)	12,600 (13,800)	14,100 (15,500)	12,900 (14,200)	9,350 (12,300)
Total	1,829 (2,017)	1,672 (1,839)	1,629 (1,793)	1,700 (1,870)	27,829 (30,559)	27,621 (30,396)	25,437 (28,553)	23,099 (27,418)
Total[3]	4,170 (4,590)	4,050 (4,460)	3,870 (4,260)	3,720 (4,090)	56,800 (62,500)	55,100 (60,600)	52,700 (58,000)	50,700 (55,800)

[1]Values from Table 7 & 8. [2]Based on projected area (Table 11) and current estimated loadings (Table 7 & 8). [3]Based on stream data(Total P and Total N).

loading to Lake Michigan is projected. Point sources were estimated to contribute about 77 percent of the loading prior to treatment (Table 4). If the agricultural contribution was about 20 percent, as estimated for the total Great Lakes Basin, this proportion would increase to about 53% when the projected 80 percent from point sources is achieved. While these values are approximations at best, it is clear that the proportion of the phosphorus loading to the Great Lakes from agricultural lands will increase as treatment is implemented for point sources. This will focus the attention on agricultural sources of phosphorus to the Great Lakes.

Because major changes in the amounts of agricultural land in the Great Lakes Basin are not anticipated, the seriousness of the problem of nutrient loadings to the Great Lakes from agricultural lands in the future, as compared to the present time, will be a function mainly of changes in practices which affect the nutrient loadings per unit area of agricultural land. In this regard, the extent to which nutrient levels in runoff are affected by fertilizer use and other agricultural practices becomes an important question.

Agricultural soils contain varying amounts of native nutrients and nutrients added as fertilizers. The large amounts of fertilizer N and P used, 260,000 and 152,000 metric tons (286,600 and 167,500 tons) respectively, for the Great Lakes Basin, have raised concern over the impact of fertilizer nutrients on loadings to the Great Lakes. Most agricultural experts (e.g. Taylor, 1967; Stanford et al., 1970; Viets 1970, 1971; Nelson, 1972) agree that insufficient information is available from sufficiently controlled situations to accurately assess the contribution of fertilizers to the nutrient loadings to lakes. Further, it is agreed that losses of fertilizer nutrients can be affected by fertilizer use practices.

The amounts of fertilizer nutrients added to the soil are small on an annual basis compared to native nutrient levels. However, over years of continuous fertilizer use, the amounts accumulated can be significant. Because of the low water solubilities of P in soils,

P is transported from soils largely in particulate form by soil erosion. Although soil NO_3-N is water soluble, soil N is largely particulate. Thus, P and N losses are closely related to the P and N contents of the soil and may be increased by fertilization of the soil. A loss of 11.2 metric tons/ha (5 tons/ac) of soil containing 1000 ppm P amounts to a loss of 11.2 kg/ha (10 lb/ac) of P. Furthermore, eroded soil is generally enriched in nutrients compared to the source soil, due to the tendency for nutrients to associate with fine silt and clay fractions. However, prediction of the impact of soil erosion on nutrient loadings to lakes, complicated by the partial deposition of sediment prior to reaching the lake and a lack of information on the availability in lakes of nutrients associated with the particulate phase (Taylor, 1967; Syers et al., 1973).

Although P and N are lost from soil mainly in eroded soil particles, the losses of these nutrients in the soluble fraction is also important. The soluble inorganic forms are directly available to aquatic plants and are more likely to reach the lake than are particulate forms. Losses of nutrients in solution are related in part to the concentrations maintained in solution by soil particles.

Fertilization is expected to increase soluble P concentrations based on soil P sorption isotherms and on the solubility of fertilizers and their reaction products in soil (Black, 1970; Ryden et al., 1973). Generally, fertilizer P reacts in soil initially to form products of relatively high water solubility (Black, 1970). As P diffuses from the fertilizer band, the concentration is decreased as P is sorbed by soil particles. As the amount of P sorbed by the soil is increased, a higher solution concentration is maintained. Consequently, both the formation of fertilizer reaction products and the increase in sorbed P would increase the tendency for loss of P in solution in runoff waters.

While some data have been obtained to show that fertilization can increase soluble P concentrations, the importance of this to P loadings to lakes has not been established. Data cited by Taylor (1967) showed that P concentrations of 13, 34 and 88 µg/l, were obtained when dilute

salt solutions were equilibrated with a silt loam soil fertilized at rates of 0, 244 and 468 kg/ha (0, 218 and 418 lb/ac), respectively. Bolton et al. (1970) obtained P concentrations of 13 and 24 µg/l in tile drainage water from non-fertilized and fertilized corn plots in a 4-year rotation with oats-alfalfa. Concentrations of 26 and 29 µg/l were obtained for non-fertilized and fertilized continuous corn plots. Nelson and Romkens (1970) observed that the P concentration in runoff water was directly related to the rate of fertilizer addition for plots receiving 0, 56 and 112 kg/ha (0, 50 and 100 lb/ac) of fertilizer P and subjected to simulated rainfall. Concentrations ranged from about 50 µg/l for the unfertilized plot to 400 for the highest level of added P. The amount of solution P lost per unit area was also directly related to fertilizer addition rate. It should be noted that the "rainfall" intensity (6.4 cm/hr or 2.5 inches/hr) and duration (1 hr) were high, contributing to a high rate of runoff. Consequently, the losses were higher than normally expected. Holt et al. (1970) reported P concentrations of 8, 9, 16 and 30 µg/l in runoff from plots receiving no fertilizer, fertilizer broadcast and plowed, fertilizer broadcast and disked, and fertilizer broadcast (rates not given), respectively.

The predominant form of N transported in solution is NO_3^-. In contrast to P, NO_3^- is not retained by the soil and moves with runoff and percolating waters. The extent of soluble N transport from the soil is a function of the rates of water percolation, nitrification, denitrification, plant uptake, and immobilization by other processes (see Section 3.4). While instances of enrichment of subsurface waters by fertilizer N are known, the importance of fertilizer N to the loadings of N from agricultural lands to lakes is not well established (Stanford et al., 1970; Viets, 1971; Keeney and Walsh, 1972).

Available information indicates that nutrients in solution contribute significantly to estimated nutrients loadings to lakes from agricultural lands (Section 3.4). Fertilization can increase concentrations of nutrients in solution and thereby increase the amounts of nutrients transported from soil in runoff and percolating waters. On this basis, future problems of nutrient additions to lakes from agricultural lands may be

reduced if emphasis is placed on balancing fertilizer application rates to agricultural crop needs and on achieving the minimum nutrient concentrations in the soil solution for maximum crop production. From another standpoint, considerable nutrient transport occurs through soil erosion and fertilization increases the nutrient content of the soil. Consequently, future problems for lakes can be reduced by avoiding excessive fertilizer use and by minimizing soil erosion.

3.9 Institutional Arrangements Relating to Great Lakes Pollution By Nutrients From Agricultural Lands

The laws, policies, and institutional arrangements pertaining to the U.S. Great Lakes Basin are discussed in detail in Appendix 20 of the Great Lakes Basin Framework Study, 1972. Apparently, programs and policies have not been developed to deal directly with pollution by nutrients from agricultural lands. However, certain agency programs and policies developed for other purposes have a bearing on nutrients from agricultural lands. Most closely related are programs concerned with control of erosion and sediment. The Fedearl Water Pollution Control Act Amendments of 1972 focused attention on diffuse sources of pollution, including sediments. The legislation called for an evaluation of the sources and extent of sediment and associated pollution arising from agricultural and urban lands, and of the legal, economic, and other implications of the implementation of erosion control methodology.

3.9.1 Agencies

The EPA, and USDA have programs concerned with agricultural pollution (EPA, 1971). The EPA is the agency with direct statutory responsibility for programs leading to abatement, prevention, and control of all water quality problems. Within the research and development program, projects are included to improve methods to alleviate or abate all agricultural sources of pollution.

Several USDA agencies are concerned with proper land use and resource conservation practices. These practices result in soil and water conservation and may result in pollution abatement, particularly sediment control. The Agricultural Stabilization and Conservation Service (ASCS) authorizes federal cost sharing with farmers for carrying out soil conservation programs and specific pollution abatement practices. Pollution abatement has been concerned mainly with animal wastes. The Agricultural Research Service conducts land and water resource research programs. Some of these relate to erosion control and pollution by agricultural chemicals. The Cooperative State Research Service administers Federal grant programs for agricultural research at state agricultural experiment stations. All experiments in the Great Lakes states are conducting research on nutrient enrichment of waters. The Soil and Water Conservation Service develops and carries out a national soil and water conservation program through soil and water conservation districts and watershed protection, flood prevention and river basin investigation projects. A survey of small watersheds in the Great Lakes St. Lawrence Region concluded that watersheds were feasible for development for water and land conservation and flood control under Public Law 566 (EPA, 1971). Local sponsoring organizations made application for 74 projects. Planning was authorized for 31 and construction for 21 projects.

Within state governments, various groups such as the Departments of Agriculture are involved in programs related to land use. Generally, these programs inform agricultural committees on farming practices, and other matters, and advise the legislative branch on subjects including agricultural pollutants.

3.9.2 Programs and Policies

Floodplain management may indirectly affect nutrient pollution from agricultural lands located in floodplains. However, management is mainly concerned with uses other than agricultural. All Great Lakes Basin states have zoning enabling legislation. Wisconsin is the only state with a compulsory floodplain zoning program (Great Lakes Basin Framework Study, 1972).

Land drainage (i.e., installation of drains, etc.) is controlled under enacted statutes in six of the Basin states (Great Lakes Basin Framework Study, 1972). Drains may be considered as public waters and subject to pollution abatement laws. Consequently, where runoff waters are channeled through drains, a potential exists for control from the standpoint of the impact of nutrient loadings to surface waters.

The states have soil conservation programs which affect pollution by nutrients from agricultural lands. All Basin states, with the possible exception of Indiana, have statutory provisions to abate or safeguard against sediment and erosion damages (Great Lakes Basin Framework Study, 1972). Erosion is an important aspect of nutrient transport from agricultural lands. Soil conservation districts may be formed in all Basin states to effect soil conservation practices, for example, as a mechanism for erosion and sediment control.

Various programs for controlling pollution from land use activities are in the planning or demonstration stages. Allen County, Indiana, Soil and Water Conservation District is working with EPA in a research and demonstration project to control pollution runoff from farmlands (Great Lakes Water Quality Board, 1973). The Minnesota Governor's Conference with participation by EPA is working to develop a model state act for soil erosion and sediment control. Michigan is considering legislation to control soil erosion. Ohio is considering a cooperative plan among state agencies to control nutrients and sediments from agricultural lands.

Under Wisconsin law, land use regulations through adopted ordinances which affect soil and water erosion, flooding, a sedimentation can be formulated by soil and water conservation districts for land use within the district but outside of incorporated villages and cities. All landowners in the district would be included under such ordinances. A model ordinance for Wisconsin soil and water conservation districts adopting land use regulations for sediment control has been developed. Wisconsin has planned a cooperative research and demonstration project to demonstrate the effectiveness of land control measures in improving water quality and to devise the necessary institutional arrangements

for the preparation, acceptance, adoption and implementation of a sediment control ordinance applicable to incorporated and unincorporated areas on a county-wide basis. The project involves cooperation between ten local, state or federal groups. Washington County in southeastern Wisconsin has been selected as the site for the proposed program.

Section 4

SUMMARY OF RELEVANT ONGOING RESEARCH DEMONSTRATION OR MONITORING PROGRAMS

4.1 Ongoing Research[1]

The nutrient contribution by the agricultural land to surface waters is difficult to quantify, because of the diffuse nature of agricultural sources. Also, the problem is complicated by a multitude of factors, both naturally occurring and man-made, influencing nutrient transport in runoff. The more obvious factors include the amount, distribution, and intensity of rainfall; the amount, quality and application of irrigation water; temperature and evapotranspiration; the amount of runoff and percolation; soil erosion and conservation practices; the physical and chemical nature of the soil; topographic, geological, and hydrological features of the earth's surface; the kinds of crops grown and the methods of crop management; and the kinds and amounts of fertilizer used and the timing of application. These factors act separately and in combination to influence the nutrient losses. Interaction among factors greatly complicates evaluation of the problem (Taylor, 1967; Stanford et al., 1970; Nelson, 1970).

Several approaches currently are used to evaluate the problem, namely studies with single or multiple watersheds, lysimeter plots, drainage plots and/or runoff plots. Monitoring of the nutrient content of streams, ponds, and lakes provides valuable data if done carefully and continuously in well selected areas where there is minimal contamination from outside sources of nutrients.

Generally, the supporting agencies are Environmental Protection Agency (Office of Research and Development), Department of the Interior (Office of Water Resources Research), Department of Agricultural (Agricultural Research Service), Department of Agriculture with Cooperative State Research Service, and State Governments.

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[1] See also Section 3.9, Institutional Arrangements.

The projects currently being undertaken are presented below according to subject matter.

4.1.1 Contribution of Nutrients from Agricultural Land to Surface Waters

Projects involving evaluation of the extent of phosphorus and nitrogen enrichment of surface waters by agricultural land are being undertaken by the following investigators: D. K. Cassell and W. C. Dahnke (North Dakota State University); G. Chesters and D. R. Keeney (University of Wisconsin); B. G. Ellis (Michigan State Univesrity) J. J. Hanway, J. M. Bremner and M. A. Tabatabai (Iowa State University); R. F. Harris and D. E. Armstrong (University of Wisconsin); H. G. Heinemann (U.S. Department of Agriculture); C. C. Hortenstine, L. C. Hammond, and R. S. Mansell (University of Florida); J. M. MacGregor and R. S. Adams (University of Minnesota); B. L. McNeal (Washington State University); L. S. Murphy (Kansas State University); D. W. Nelson and M. J. Romkens (Purdue University); R. A. Olson, J. Muir and E. C. Seims (University of Nebraska); B. L. Schmidt and T. J. Logan (Ohio Agricultural Research and Development Center); D. R. Timmons and R. F. Holt (U.S. Department of Agriculture); E. M. White (South Dakota State University); and P. J. Zwerman, D. J. Lathwell and D. R. Bouldin (State University of New York at Ithaca). One investigator, G. O. Schwab (Ohio State University) is studying specifically the movement of nutrients from agricultural land to Lake Erie.

4.1.2 Contribution of Nutrients by Fertilizers to Surface Waters

Current projects on the role of fertilizers in pollution of surface waters include those conducted by: R. R. Bradford (Alabama Agricultural and Mechanical College); L. A. Douglas (Rutgers, the State Univesrity); E. P. Dunigan (Louisiana State University); J. E. Giddens (University of Georgia); J. W. Gilliam (Univesrity of North Carolina);

J. W. Gilliam and S. B. Weed (University of North Carolina); G. F. Griffin and R. W. Wengel (University of Connecticut); J. J. Hanway, R. M. Shibles, and E. J. Dunphy (Iowa State University); L. H. Hileman (University of Arkansas); D. M. Himmelblau and M. Hildebrand (University of Texas); C. C. Hortenstine, D. A. Graetz and D. F. Rothwell (Univesity of Florida); D. E. Kissel and C. W. Richardson (Blackland Conservation Experiment Station); W. Kroontje (Virginia Polytechnical Institute); G. M. Lessman (University of Tennessee); J. F. Lutz (University of North Carolina); T. M. McCalla and G. L. Schuman (University of Nebraska); J. D. Menzies and G. Stanford (U.S. Department of Agriculture); R. A. Olson, A. D. Flowerday, D. Knudsen and G. A. Peterson (University of Nebraska); A. R. Overman (University of Florida); T. C. Peele (Clemson University); C. E. Scarsbrook (Auburn University); G. E. Smith (University of Missouri); A. Swoboda (Texas A & M University System); G. W. Thomas (University of Kentucky); M. R. Till and D. W. Armstrong (South Australian Department of Agriculture); F. G. Viets and S. R. Olsen (U.S. Department of Agriculture); L. F. Welch, J. D. Alexander and T. E. Larson (University of Illinois); C. W. Wendt, A. B. Onken and O. C. Wilke (South Plains Research and Experiment Center); and W. M. Winant (university of Vermont).

4.1.3 Nitrogen and Nitrates in the Environment

Nitrogen and nitrates in teh soil, plants and surface waters are under investigation by: M. M. Alexander and P. L. Minott (State University of New York); S. A. Barber and D. W. Nelson (Purdue University); A. V. Barker and D. N. Maynard (University of Massachusetts); J. M. Bremner and L. G. Bundy (Iowa State University); J. C. Day (Univesrity of Arizona); Y. Kanehiro (University of Hawaii); D. R. Keeney (Univesrity of Wisconsin); F. E. Koehler and N. K. Whittlesey (Washington State Univesrity); W. D. Lembke, J. K. Mitchell and J. Simon (Unviversity of Illinois); G. D. Lewis, D. V. Naylor and D. W. Fitzsimmons (University of Idaho); A. D. McLaren and R. K. Schulz (University of California); R. J. Miller (University of California); D. W. Nelson (Purdue University); N. K. Peterson

(University of New Hampshire); R. V. Rourke and R. F. Stafford (University of Maine); J. R. Sims (Montana State University); E. R. Swanson (University of Illinois); A. R. Swoboda (Texas A & M University System); J. M. Tiedje (Michigan State University); S. J. Toth (Rutgers, The State Univesrity); T. C. Tucker, G. R. Dutt and R. L. Westerman (Univesrity of Arizona); R. A. Young and R. J. Ruf (University of Nevada); V. V. Volk and M. G. Cropsey (Oregon State University); C. W. Wendt (Texas A & M University System); and F. Wiersma (University of Arizona).

Section 5

NATURE AND AVAILABILITY OF TECHNOLOGY TO COPE WITH NUTRIENTS IN AGRICULTURAL RUNOFF

Soil erosion and sediment transport play a major role in the transport of nutrients in agricultural runoff (Taylor, 1967; EPA, 1971; Sections 3.4 and 3.5). Sediment transport has long been recognized as a problem in agricultural areas, and soil conservation and sediment control practices have received considerable attention (Glymph and Storey, 1967; Amemiya, 1970; EPA, 1971). Generally, technology to control sediment transport is available, but further effort is needed in implementation and development of practices most compatible with intensive agricultural production. Practices used in controlling erosion and sediment transport include the following: 1) cover crop maintenance, 2) strip cropping, 3) crop rotation, 4) contour farming, 5) mulching, 6) mulching with minimum tillage, 7) rough tillage to increase infiltration, and 8) terracing and other slope alteration practices. Maintenance of a cover crop or crop residue cover on the soil is recognized to be of major importance in soil stabilization and erosion control.

Soil fertilization increases the available nutrient supply for crop utilization and for transport in surface runoff (Taylor, 1967; Stanford et al., 1970; EPA, 1971). While instances of increased nutrient concentrations in runoff from fertilized soils have been reported (see Section 3.8), the impact of soil fertilization on nutrient transport in runoff is uncertain.

Aside from sediment transport, the movement of soil phosphorus in surface runoff should be related to the concentrations of soluble phosphorus maintained in the soil solution. Consequently, the forms and amounts of phosphorus applied, the timing of phosphorus applications in relation to expected periods of high runoff, and the incorporation of fertilizer phosphorus into the soil are important in minimizing phosphorus transport in runoff (Taylor, 1967; Martin et al., 1970).

The solubility of NO_3^- has focused major attention on the mobility of NO_3^- in surface runoff and percolating waters. Fertilization practices which minimize NO_3^- losses have been emphasized. Important practices include balancing application rate with crop needs, split applications and side-dressing to time applications with crop utilization, and use of deep-rooted crops in crop rotations to return leached NO_3^- to the soil surface (Martin et al., 1970; Stanford et al., 1970). Nitrification inhibitors may prove useful in retarding NO_3^- formation and thereby reducing losses.

Practices which may lead to increased transport of nutrients from agricultural soils include fall fertilizer application, broadcast application, and addition of fertilizers, including manure, to frozen soil (Biggar and Corey, 1969; Martin et al., 1970). Relatedly freezing apparently enhances the release of nutrients from crop plants, and this process may contribute substantially to nutrient transport during the spring runoff period (Holt, 1969).

The concern over water quality deterioration, the increasing costs and decreasing supplies of fertilizers, and decreasing food supplies emphasize the need for technology to maximize efficient fertilizer use and crop production while minimizing water pollution problems. Procedures for evaluating fertilizer application rates should not only focus on maximizing production but on minimizing fertilizer use and the potential for loss in runoff and seepage waters.

Section 6

NEED FOR NEW RESEARCH, DEMONSTRATION OR MONITORING PROGRAMS

6.1 Gaps in Knowledge Likely to be Remaining By the 1976 Time Frame

Accurate estimates fo the nutrient contribution of agricultural lands to the Great Lakes can not be made at the present time. The contribution is estimated to be about 20 percent of the total loading for phosphorus and will increase in proportion as point sources of phosphorus pollution are reduced through waste treatment. However, information on nutrient transport from agricultural land has been obtained from widely scattered locations and frequently under poorly defined conditions. Because of the lack of precision in estimates of nutrient transport in runoff, accurate evaluations of the role of land use and other factors in nutrient loadings can not be made.

The factors and mechanisms controlling the amounts of nutrients transported from agricultural lands are not established sufficiently to facilitate management programs. The physical and chemical factors controlling concentrations of soluble nutrients contained in surface runoff are not understood to the extent that quantitative predictions can be made. The effect of fertilizer use on soluble nutrient transport is uncertain.

Information is lacking on nutrient transport in streams and rivers. It is evident that a considerable portion of the soil eroded from agricultural lands is deposited prior to discharge into lakes. Furthermore, soluble nutrient loss from runoff waters through interaction with stream bank and bottom sediments can not be predicted.

The effect of particulate nutrients transported to lakes on the nutrient status of lake surface waters is poorly understood. Consequently, the validity of including particulate nutrients in lake nutrient budgets is uncertain. This arises because of a lack of information on the rates of release of nutrients from particulate forms in lake systems, the

availability of particulate forms to aquatic organisms, and the rates of nutrient uptake from particulate forms by algae as related to the longevity of particulate nutrient forms in lake surface waters.

6.2 New Studies Needed to Define the Problems Remaining

Gaps in knowledge regarding nutrient transport from agricultural lands to the Great Lakes (Section 6.1) indicates a need for research and demonstration projects in the areas discussed below to provide the information required.

6.2.1 Basic Research

Intensive investigation and monitoring of nutrient transport from representative agricultural watersheds. This type of investigation is needed to provide quantitative information on nutrient transport as a function of land use, soil type, fertilization, runoff and other important factors. Measurements should be made over a sufficient time period to provide a representative, accurate assessment. Relation to important events such as spring runoff should be evaluated to facilitate management and control programs.

Determination of the factors and mechanisms controlling the amounts of nutrients transported from agricultural lands. Research on the physical and chemical factors controlling the concentrations of soluble nutrients in runoff should be emphasized. The effects of soil properties, fertilizer application, and fertilization history of the soil should be established. This information would identify the specific processes and factors controlling soluble nutrient transport and, in turn, aid programs to control the problem.

Determination of the factors and processes controlling nutrient transport in streams and rivers. The extent of soluble nutrient removal by stream bank and bottom sediments should be established. This information would relate measurements of nutrient transport from agricultural land to the amounts expected to reach the Great Lakes.

Determination of the effect of particulate nutrients in runoff on the nutrient status of lake surface waters. Information is needed on a) the rate and extent of nutrient release from particulate forms, b) the biological availability of particulate nutrient forms and c) the rates of utilization of particulate nutrients as related to the rates of particulate nutrient removal from lake surface waters through settling. This information would greatly improve predicting of the impact of agricultural runoff on lake water quality.

6.2.2 Demonstration Projects

Monitoring of representative agricultural drainage basins. As discussed above (Section 6.2.1) this type of program is needed to provide quantitative information on nutrient transport from agricultural lands.

Effectiveness of sediment control through soil and water conservation programs in reducing the amounts of nutrients transported from agricultural lands. Considerable amounts of nutrients are transported with eroded soil. Control of soil erosion will also reduce the amounts of soluble nutrients transported from soil. Projects such as those described for Allen County, Indiana and Washington County, Wisconsin (See Section 3.9) are needed to determine whether soil conservation and sediment control programs are effective in reducing sufficiently the transport of nutrients from agricultural lands, and whether these programs can be implemented under existing legislation, policy and institutional arrangements.

6.2.3 Development of New Technology

It is likely that the basic technology exists to control nutrient transport from most agricultural lands through appropriate farming practices and soil erosion control programs. Furthermore, it is likely that technology can and will improve. However, the research and demonstration projects described above are needed to establish sufficiency of existing technology. If this technology is found to be adequate, the

major problem remaining is implementing the appropriate technology throughout the Basin. Since agriculture is likely to become more intensive to increase food production per unit area, fertilizer use technology should receive careful attention. Excessive fertilizer use will become less economical and may be detrimental to water quality. Methods of assessing fertilizer needs should be examined closely to ensure that these methods predict the appropriate time, rate, type and method of fertilizer application so that a high level of production can be maintained with minimal harmful effects on water quality.

Section 7

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PRINCIPAL REFERENCES

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Category A7 - Erosion and Sedimentation

ASSESSMENT OF EROSION AND SEDIMENTATION TO THE U. S. PORTION OF THE GREAT LAKES BASIN

prepared by
THE UNITED STATES SOIL CONSERVATION SERVICE, U.S.D.A
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GREAT LAKES BASIN COMMISSION
Ann Arbor, Michigan

To be used as portion of the U. S. Task A Report on GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by the International Joint Commission - prepared in partial fulfillment of U. S. Environmental Protection Agency Contract No. 68-01-1598

June 4, 1974

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Section I-1

INTRODUCTION

In this paper, sediment to the Lakes is the product of sheet and gully erosion of the soils from the agricultural and urban land use categories only. The larger sediment yield rates measured on the major streams in the basin are 87 metric tons per square kilometer (250 tons per square mile) from the Cuyahoga River at Independence, Ohio and 263 metric tons per square kilometer (750 tons per square mile) from the Genesee River in New York. While these sediment yields are moderately low in comparison to other parts of the U. S., sediment yields of this magnitude can cause serious problmes to harbors and fish. The sediment is a carrier of nutrient elements and pesticides. It also creates turbidity in the streams and the Great Lakes.

Erosion creates a wide range of sediment sizes ranging from coarse to fine grained material. Particles of all size fractions no doubt are contributed as sediment yield to the Lakes. The major portion of the sediment, however, is fine grained.

The most severe erosion occurs around the southern end of Lake Michigan and south of Lakes Erie and Ontario. Rivers and streams which drain this area reflect these higher erosion rates by a corresponding higher sediment yield than in other streams in the basin.

Section I-2

SUMMARY OF FINDINGS AND CONCLUSIONS

Sediment yield to the Lakes from erosion on agricultural and urban land from the U. S. portion of the Basin can be estimated with reasonable accuracy. Wise land use and effective use of soil conservation systems can effectively reduce or minimize these sedimentation problems. Conservation practices which are a part of these systems reduce runoff and erosion. If the conservation program for the future is of similar magnitude to that of the past, then the sediment yield to the Lakes is expected to remain about the same as present until 1980 and decline slightly below present levels in 2000. With present rates of land treatment by 2020 the sediment yield is expected to increase to about 6% above the present figure. This variation in yield is due mainly to an expected decline and then increased demand for row crops in the Basin. The general upward trend in yield is attributed to the projected continuing increase in areas undergoing urban construction. If the conservation program is increased above the present rates, then a significant decrease in sediment yield can be expected. The amount of decrease will be directly related to the amount of soil conservation practices which can be installed.

If the estimates developed in this paper are to be refined, new studies will be needed. These studies will be in the nature of research, data gathering, and an expanded water quality monitoring program.

The research needs are quite broad and cover items such as refinement of sediment delivery ratios for large watersheds, effect of land treatment on sediment yield from large watersheds and refinement of the factors influencing erosion and sediment yield from urban land. The data gathering should include remeasuring of sediment in reservoirs already measured and the measurement of new reservoirs. Expanding the water quality monitoring program would be necessary. The addition of more sediment gauging stations in the program for an adequate period of time would provide the data to predict sediment yield to the Lakes more accurately. These stations will also provide data on the effect of changes in land use on sediment yield.

Section I-3

RECOMMENDATIONS

Accurate and reliable estimates are needed on the sediment yield to the Lakes. Additional stream gauging stations where sediment is measured are required. Many more stations located near the junction of the stream and lake are needed to measure total suspended load from the streams flowing directly into the lake. Additional stations on the streams within the watershed are required to provide data with which sediment yield from different land uses can be determined.

One of the basic procedures used in sedimentation work utilizes sediment delivery ratio as a technique to reduce gross erosion to sediment yield. The sediment delivery ratio curve now in use was not developed for use on large watersheds. Research should be done to develop a new curve which is valid for large watersheds.

It is desirable to start a program to measure sediment in all reservoirs which meet the requirements for this type of operation. Reservoirs which have not been measured as well as those measured previously should be brought under this program. These data will help refine (1) sediment yield from large watersheds, (2) sediment delivery ratio, (3) the effect of land treatment on sediment yield on large watersheds, (4) sediment yield from and the effect of land treatment on soils undergoing urban construction and (5) sediment yield from settled urban areas.

Research should be intensified to determine the magnitude of sediment yield from urban land and from soils undergoing urban construction. Material gathered from this study can be used to refine procedures for predicting sediment yield from these types of areas.

Ordinances and regulations should be developed in each state on the control of erosion, runoff and sediment. These ordinances should be adequately funded so that their operation and enforcement is insured.

There should be a program which would offer the farmer and urban developer an incentive to apply sediment control measures. This could be in the form of cost-sharing or tax write-off.

Section II-1

INTRODUCTION

Pollution of the Great Lakes by sediment has been going on since the Lakes were formed. This sediment was from what is considered normal or geologic erosion. It was only when man manipulated the vegetation and land use for his own ends that the erosion on the land and sediment yield to the Lakes made a significant increase. Since studies of pollution to the Lakes began, it has been recognized that agricultural sediment, from the standpoint of bulk, is the major pollutant to the Lakes. However, to simply brand it as the most serious pollutant because of quantity may be an erroneous conclusion.

Sediment yield to the Great Lakes was estimated to be 453,900 metric tons(500,000 short tons) per year in the early 1800's. This yield has now increased to about 4,316,200 metric tons (4,757,800 short tons) and unless an intensive land treatment program is instituted the sediment yield is expected to increase to 4,557,100 metric tons (5,623,000 short tons) by 2020. Sediment yields of this magnitude, while larger than they should be for the intensity of development which exists, are not a serious problem to the Great Lakes as a whole. Serious local problems exist where streams discharge sediment into the Lakes. Sediment at these locations deposits in harbors, destroys fish habitat, reduces the recreational value of the lake, and increases municipal water treatment costs.

Section II-2

SCOPE OF STUDY

The details of this study cover sediment from sheet and gully erosion on agricultural land and urban areas but not from streambank erosion, forested land, transportation facilities, mining property, or recreation land. Information on sediment from these categories will be located in their respective reports. Originally the sediment assessment from urban land was to be the point where sediment entered sewers or man-made conveyances. This type of breakdown was not possible within the time allotted so this report includes all sediment from sheet and gully erosion on urban land. The material considered for this study covered a wide range, all the way from general reports on sedimentation to papers on specific subjects within the field.

There were several sources of data which were considered. They include files of the Soil Conservation Service, five Annotated Bibliographies on Hydrology and Sedimentation by the Water Resources Council or its predecessor plus one by the USGS, Catalog of Information on Water Data, two Inventories of Sediment-Load Data, a Summary of Reservoir Sediment Deposition Surveys, 15 issues of Notes on Sedimentation Activities, abstracts from the Smithsonian Science Information Exchange and from the Current Research Information System. Specific sources are detailed in Section II-3.

Sedimentation as a science is a relatively young discipline. Active interest in the field started in the early 1930's and has been growing ever since that time. It was just this year (1973) that the American Society of Testing Materials instituted a committee on this subject. Since there has been an increasing interest and study of this discipline, tremendous progress has been made in predictive techniques and measurement of sediment. Because of the dynamic progress that has overtaken the discipline, few papers or studies before 1950 were considered germane to this report. In most cases data developed before 1950 have been refined and expanded to the point where it would be redundant and possibly inappropriate to use the older material. Therefore, no material referring specifically to the Great Lakes was used which was dated before 1950. However, a book by H. H. Bennett of 1939 vintage was included in the general reference material.

Section II-3

STUDY PROCEDURE

The study was accomplished by first conducting a literature review. After this review contacts were made with various research personnel for interviews. The list of interviews does not necessarily represent all individuals working on these "categories" who could contribute data. It really represents those individuals which because of time and budget limitations could be contacted during the course of normal work.

Interviews were held with:

Ohio State University
- Agronomy Department
  - Dr. L. P. Wilding
  - Dr. T. Logan
- Agricultural Engineering Department
  - Dr. G. O. Schwab

University of Cincinnati
- Geology Department
  - Dr. P. E. Potter

Purdue University
- Agronomy Department
  - Dr. J. V. Mannering
  - Dr. D. Wiersma
- Agricultural Engineering Department
  - Dr. E. J. Monke

Agricultural Research Service
- W. H. Wischmeier, Research Statistician

Soil Conservation Service
- Lansing, Michigan
  - J. Thompson, Geologist
- Madison, Wisconsin
  - P. Cavanaugh, River Basin Party Leader
  - R. N. Cheetham, Jr., Geologist
- Columbus, Ohio
  - J. H. Harrington, Jr., State Conservation Engineer
- Indianapolis, Indiana
  - E. Pope, State Conservation Engineer
  - L. Kimberlin, State Resource Conservationist
  - C. Gossett, Geologist

Sources of literature which were reviewed are as follows:

1. Files of the Soil Conservation Service

2. Annotated Bibliography on Hydrology 1951-1954 and Sedimentation 1950-1954. Bulletin No. 7, Dec. 1955, Subcommittee on Hydrology and Sedimentation - Inter-Agency Committee on Water Resources.

3. Annotated Bibliography on Hydrology and Sedimentation 1959-1962 (U. S. and Canada). Bulletin No. 8, Sept. 1964, Subcommittees on Hydrology and Sedimentation - Inter-Agency Committee on Water Resources.

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Section III-1

GENERAL DESCRIPTION OF LAND USE ACTIVITY

Sediment sources have been classified according to causative factors, eroding agent, location and other criteria. Thus, there are man-made (accelerated) and natural geologic erosion; water, wind, ice and gravity erosion; and erosion of the land surface (sheet-rill) or drainageway (gully-channel) erosion. For the Great Lakes Basin, the predominant sediment sources are from sheet-rill erosion of cultivated soils.

Agricultural land, because of type and intensity of use and because of the large area exposed and its susceptibility to erosion supplies the greatest amount of sediment yield to the Great Lakes. Approximately 80 percent of the sediment delivered to the Great Lakes comes from sheet, rill and gully erosion on agricultural and urban land. The remainder is from sheet and gully erosion on forest land, from road banks, stream banks, and municipal and industrial wastes.

The most widespread influence, other than on site land damage from the erosion process, is the downstream effect on water and related land resources. Runoff water carries the product of erosion, and other solid wastes, and deposits this product as sediment. Sometimes this product remains suspended in the downstream waters for prolonged periods and constitutes a detrimental element to the quality of water.

Section III-2

LENGTH OF TIME ACTIVITIES IN PROGRESS

Erosion and the resultant sediment yield are natural processes and as such have been continually active. Before settlement of the land, most of the streams in the basin probably ran clear except for periods of flood and high water. Accelerated erosion no doubt started in the middle 1800's with land settlement, land clearing, and with shifting much of this land into farms (22)[1/]. Land settlement combined with poor farming methods and urban construction practices started the trend which has continued until recently of an ever increasing sediment yield to the lakes.

The discussion of the magnitude of erosion and sediment yields to the Lakes during this early period is mostly conjecture because little data on this subject were obtained until fairly recently. Even today there are insufficient data to make more than estimates of the problem. However, the beginning of the accelerated erosion process could not have been before land settlement and the farm population by 1850 was such that had records of soil erosion and sediment yield been kept they certainly would have recorded a noticeable change from the period before 1800. This has been verified by Kemp using pollen variations in recent lake sediments (22).

Since the premise has been made that accelerated erosion and sediment yield to the Lakes is related to the amount of land used as cropland, the following table might be of interest (12).

Date	Cropland (U. S. Portion of the Basin)	
	(millions of hectares)	(millions of acres)
1800	-	-
1850	1.8	4.4
1920	5.3	13.0
1970	13.0	32.1

1/ Numbers in parentheses refer to references cited in section VII, Principal References.

Section III-3

TYPES AND NATURE OF POLLUTANTS ASSOCIATED WITH SEDIMENT

A large proportion of pesticides, plant nutrients, nuclear fallout and infectious particles leave the land and enter water bodies by attachment to sediment particles.

The amount of nitrogen, phosphorus, and potassium reaching surface waters that can be attributed to the fertilization of agricultural lands are yet unknown. Not all nutrients in waters come from fertilizers. Large amounts come from decomposition of soil organic matter and other sources. Little phosphorus can be lost from most soils except by erosion. Nitrogen may be in organic compounds carried by sediment, from decaying plants which might have gotten the nitrogen from fertilizers, or the atmosphere.

Levels of pesticides found to move from agricultural lands may be found in varying amounts in the runoff water and/or sediments. For example, studies by the Agricultural Research Service show that the amount of DDT residue in soils was related to the amount of DDT applied in previous years. The amount of DDT dissolved in runoff water was not detectable but significant levels were found adsorbed on surfaces of sediment particles.

Section III-4

STATE OF THE ART

There are several ways to determine the sediment yield of a watershed depending upon the physical features and the data available (30). Average annual sediment yields may be obtained from: (1) sediment load records; (2) gross erosion and sediment delivery ratios; (3) measured sediment accumulation; and (4) predictive equations.

Suspended load records.--The most reliable method of determining sediment yield from large watersheds. Suspended sediment transported by a stream may be measured by sampling. Water discharge can be determined by gaging at particular stream cross sections. Sediment yields may be estimated from these data. When field data are obtained in a proper manner for an appropriate period of time, the calculated average annual suspended sediment yield can be considered quite reliable.

There are literally hundreds of streams draining into the Great Lakes but unfortunately only a few have stations where suspended load is measured. Of the 1329 stations recorded only 60 have sediment data (10)(33). Of these 60, most of them, or 48, have a record of over 5 years but only 12 have a record over 10 years. Of those stations with over 10 years record only 3 have weekly or daily records. Table 1 is an inventory of all sediment stations on streams in the Great Lakes Basin showing the name of station, location, and type of information acquired (10)(33).

In order to estimate suspended load to the Lakes the stations should be as close to the mouth of the stream as technically feasible so as to keep unmeasured flows below the station to a minimum. Many of the above mentioned stations are not located at the mouths of streams but are well up in the watershed. They were not established for the primary purpose of estimating suspended sediment yield to the Lakes.

Suspended load is only part of the total load moving in a stream. The other portions are bed load and sediment attached to floating material. These items are considerably more difficult to measure. Bed load often is estimated as a percentage of suspended load.

Gross erosion and sediment delivery ratios --- This method has been used with success by the Soil Conservation Service for many years (30). It is well suited for estimating current sediment yields and predicting the effect of land treatment and other measures on future sediment yields. The estimate of sediment yield is made by use of the following equation:

$$Y = E(DR)$$

where Y = sediment yield (tons/unit area/year)
E = gross erosion (tons/unit area/year)
DR = sediment delivery ratio (DR less than 1)

The gross or total erosion is the summation of all the water erosion taking place. It is developed by the use of Universal Soil Loss Equation or other methods. The sediment delivery ratio is selected from various curves after considering the following: (1) type of sediment sources; (2) magnitude and proximity of sediment sources; (3) the transport system; (4) texture of eroded material; (5) depositional areas enroute from source to point of consideration; and (6) watershed characteristics. The product of gross erosion and sediment delivery ratio is the sediment yield.

While this procedure has worked well in the past, it has been developed and used on watersheds of less than 1,036 square kilometers (400 sq. miles). The delivery ratio curve is only plotted for watersheds up to 1,554 square kilometers (600 sq. miles). The use of this technique for catchments such as the Maumee River of about 3,315 square kilometers (1,280 sq. miles) and The Grande River of about 14,660 square kilometers (5,660 sq. miles) by extending the curve might be expanding the curve past its limits of accuracy. Also the soil loss equations were not developed for prediction on large areas of less than 1 percent slope. However, the data developed by this technique has checked reasonably well with the few suspended load stations.

Measured sediment accumulation -- The measured sediment accumulation in reservoirs of known age and history are excellent sources of data for establishing sediment yields (30). However, reservoir deposition and sediment yield are not synonymous. The amount of accumulated sediment must be divided by the reservoir's trap efficiency to obtain the sediment yield. This takes into account the amount of sediment that passed through the reservoir.

The sediment yield of an unmeasured watershed may be estimated from that of a measured watershed in an area where the topography, soils, and land use are similar. In order to directly transpose sediment yield data, the size of the drainage area of the surveyed reservoir should not be less than one-half nor more than twice that of the watershed under consideration. Beyond these limitations the annual sediment yield may be adjusted on the basis of curves and formulas after considering the same factors listed under the delivery ratio discussion.

Unfortunately only 52 reservoirs have been surveyed in the Great Lakes Basin and none of them have watersheds of a size comparable to the major stream systems (5). As in the discussion of suspended load the formulas and curves were developed for catchments of less than 1,036 square kilometers (400 sq. miles). Extrapolating this data to watersheds the size of major streams in the Great Lake Basin might be overextending the accuracy of the curves and formulas. Table 2 is a list of all reservoirs surveyed for sediment in the Great Lakes Basin showing name

of reservoir, location, date of survey, drainage area, storage capacity, and total and average sediment accumulation.

Predictive equations -- Predictive equations based on watershed parameters have been developed in some areas to estimate sediment yield (30). These equations express sediment yield as a function of a combination of several measurable, independent variables. The variables may be the size of the drainage area, annual runoff, watershed shape, relief-length ratio, average slope, and expression of the particle size of the surface soil and others.

Such equations are not numerous but, where developed, they can be used with the understanding that their application must be confined to the specific area they represent. All formulas of this type so far have been developed in the western United States. Since none of these formulas are known to have been developed which are applicable to the Great Lakes Basin this approach does not appear viable.

A review of the methods used to predict or determine sediment yield would be incomplete without a discussion of erosion and problems of determining its magnitude.

Investigations have identified the basic factors involved in the sheet erosion process to be rainfall, soil erodibility, slope length, slope gradient, kind and condition of cover and an erosion control factor such as contouring. These factors have been incorporated into equations that provide a means of obtaining a quantitative estimate of the amount of soil material moved by the sheet erosion process. The basic procedure and weaknesses as applied to the Great Lakes Basin were mentioned previously under gross erosion and sediment delivery ratios.

The factors listed above most susceptible to change by man are kind and condition of soil cover and slope length. When the kind or condition of the soil cover is improved such as using a cover crop rather than leaving the land bare or the slope length decreased by means of terraces less soil is eroded. A quantitative estimate of the change can be determined by use of the equations. Information obtained in this manner is normally from small catchments because the factors necessary for use in the equation are not usually available for the larger basins. Also, while we know that structural measures, which can effect slope length, can cause a reduction of erosion we do not have precise techniques for predicting the exact amount of this reduction from large areas.

Gullies usually follow rill erosion, beginning in slight depressions of the land surface where in time the concentrated flow may cut a considerable channel. The shape of the channel is generally influenced by the relative resistance of the soil or underlying rock.

Streambank erosion and bed degradation are influenced primarily by the bank materials and resistance of the channel bottom to the character and direction of flow. Removal of natural vegetation from streambanks increases bank erosion. Increases in peak flows or prolonged above normal flows due to changes on the watershed such as urbanization can increase bank erosion. The presence of coarse bed material that a stream cannot pick up during reduced flows tend to result in the banks being attacked by the flowing water.

The determination of loss by the various types of channel erosion can be made in several ways such as (1) a comparison of aerial photographs of different dates to determine the annual growth of channels; (2) re-running existing cross sections to determine the difference in total cross-sectional area; (3) the assembly of historical data in order to determine the average age of channels and their average annual growth; and (4) field studies to estimate the average annual rate of growth, either by lateral erosion or incision, in terms of volume per unit length of channel.

Again these techniques are more appropriate to small watersheds than to large basins as the computations could tend to be cumbersome and unwieldy for large areas.

Sediment delivery ratio is an important fluvial sediment concept that can be defined as the ratio of the amount of sediment carried out of a basin to the gross erosion within the basin. As has been mentioned before reliable techniques for determining sediment delivery ratio have not been developed for large basins.

Once the process of transportation of sediment starts interest is usually on the mechanical process of movement or entrainment. A facet of entrainment that is generally neglected is the chemical changes in the clay fraction that could or might take place (32).

These changes could markedly affect the ability of the clay to carry fertilizer elements. Biological alteration of inorganic compounds could also take place (32). Data of this type are generally lacking.

The state of the art of estimating quantities of erosion resulting from urban areas and from areas undergoing development is in its infancy. This is equally true of estimating how much of the eorded material goes downstream and how far.

The best approach to determining sediment yield from urban and urbanizing areas is to measure the sediment either as suspended load or in a reservoir. Unfortunately there are few gauging stations or reservoirs located on small watersheds where urban sediment can be measured.

Another method is to use the gross erosion and sediment delivery ratio technique. Watersheds under urban development usually are small enough that size is not a limitation. However, potential soil loss equations were not developed with urban criteria so modification of the equations could introduce an error.

A topic which has not been didcussed is the availability of soils information. This information is in Published Soil Surveys. Published Soil Surveys contain a wealth of data which is essential for making predictions on erosion, sediment yield and planning for conservation treatment. Soil series and phase, and land capability classification are but a few of the items available in the report which are useful to the individual working in this discipline.

The attached map of the Annual Status of Soil Surveys shows that mapping is completed or in progress on approximately 98% of the U. S. portion of the Basin and that emphasis for completion is given to the most intensively used areas. It also shows that mapping is less than 50% complete in about 60% of the Basin. For small detailed sedimentation studies where published soil surveys are not available personal interviews with the soil survey party leader can probably furnish sufficient data. It would not be practicable to interview all the individuals necessary for a large broad general study. In this case it would be most desirable to have Published Soil Surveys available. The latest estimate on completion of soil mapping in the Basin is 1995 with present funds, or 1990 with additional funds.

Soils information with a less intensive approach than that contained in the Soil Survey Report can be obtained from a General Soil Map. Such maps have been completed for all states in the U. S. portion of the Basin. General Soil Maps are also available for most counties.

Table 1 - Sampling Stations with Sediment Data

MAP NUMBER	MAP LETTER	OWDC NUMBER	AGENCY STATION NUMBER	STATION NAME	LATITUDE	LONGITUDE	STATE	COUNTY	SITE	PERIOD OF RECORD BEGAN	PERIOD OF RECORD DISCONTINUED	INTERRUPTED RECORD	SEDIMENT: CONCENTRATION (SUSPENDED)	SEDIMENT: PARTICLE SIZE (SUSPENDED)	SEDIMENT: PARTICLE SIZE (BED MATERIAL)	SEDIMENT: OTHER	AGENCY REPORTING
				GREAT LAKES REGION													
21	Y	50509	04201500	ROCKY R NR BEREA OHIO	412422	0815313	OH	035	STREAM	1964			4	8	8		GS
21	Y	50508	04200500	BLACK R AT ELYRIA OHIO	412250	0820615	OH	093	STREAM	1962			8	8	8		GS
21	Z	50517	04208000	CUYAHOGA R AT INDEPENDENCE OHIO	412344	0813745	OH	035	STREAM	1948			3	8	8		GS
21	AA	50523	04212500	ASHTABULA R NR ASHTABULA OHIO	415119	0804543	OH	007	STREAM	1965			8	8	8		GS
21	AA	50521	04212000	GRAND RIVER NR MADISON OHIO	414426	0810248	OH	085	STREAM	1965			8	8	8		GS
21	AA	50518	04209000	CHAGRIN R AT WILLOUGHBY OHIO	413751	0812413	OH	085	STREAM	1965			4	8	8		GS
21	AB	50524	04213000	CONNEAUT C AT CONNEAUT OHIO	415534	0803618	OH	007	STREAM	1965		*	8	8	8		GS
21	AC	50507	04199500	VERMILION R NR VERMILION OHIO	412255	0821900	OH	093	STREAM	1950		*	8	8	8		GS
21	AC	50506	04199000	HURON R AT MILAN OHIO	411800	0823630	OH	043	STREAM	1950		*	8	8	8		GS
22	B	51813	04100500	ELKHART R AT GOSHEN IND	413536	0855055	IN	039	STREAM	1963	1968		*	*			GS
22	O	54386	04142000	RIFLE R NR STERLING MI	440421	0840112	MI	011	STREAM	1966	1968		4	8			GS
22	Q	54389	04151500	CASS R AT FRANKENMUTH MICH	431950	0834525	MI	145	STREAM	1966			4	8			GS
22	T	54388	04144500	SHIAWASSEE R AT OWOSSO MICH	430010	0841115	MI	155	STREAM	1966			4	8			GS
22	W	54390	04159500	BLACK R NR FARGO MICH	430532	0823705	MI	147	STREAM	1966	1970		4	8			GS
22	X	54392	04164000	CLINTON R NR FRASER MICH	423440	0825700	MI	099	STREAM	1966	1970		4	8			GS
22	Y	55081	32903 04165700	DETROIT R AT DETROIT MICH	422050	0825731	MI	163	STREAM	1957			*				FPA
22	AB	54393	04176500	R RAISIN NR MONROE MICH	415750	0833155	MI	115	STREAM	1966			4	8			GS
22	AE	51814	04182000	ST MARYS R NR FORT WAYNE IND	405916	0850603	IN	003	STREAM	1953	1968		*	*			GS
22	AF	50883	04185000	TIFFIN R AT STRYKER OHIO	413015	0842550	OH	171	STREAM	1952		*	8	8	8		GS
22	AG	50887	04189000	BLANCHARD R NR FINDLAY OHIO	410321	0834117	OH	063	STREAM	1965			8	8	8		GS
22	AH	50889	04191500	AUGLAIZE R NR DEFIANCE OHIO	411415	0842357	OH	039	STREAM	1965			8	8	8		GS
22	AH	50885	04186500	AUGLAIZE R NR FT JENNINGS OHIO	405655	0841558	OH	137	STREAM	1965			8	8	8		GS
22	AI	50892	04193500	MAUMEE R AT WATERVILLE OHIO	413000	0834246	OH	095	STREAM	1950			3	8	8	*	GS
22	AK	50901	04197000	SANDUSKY R NR MEXICO OHIO	410239	0831142	OH	147	STREAM	1965		*	8	8	8		GS
22	AK	50900	04196800	TYMOCHTEE C AT CRAWFORD OHIO	405520	0832100	OH	175	STREAM	1965			8	8	8		GS
22	AK	50897	04196000	SANDUSKY R NR BUCYRUS OHIO	404813	0830024	OH	033	STREAM	1965			8	8	8		GS
23	K	51203	04063700	POPPLE R NR FENCE WIS	454550	0882750	WI	037	STREAM	1963			4	7			GS
23	K	51202	04061000	BRULE R NR FLORENCE WIS	455331	0881557	MI	071	STREAM	1964	1968		8				GS
23	K	51204	04066000	MENOMINEE R NR PEMBINE WIS	453525	0874635	MI	109	STREAM	1964			*				GS
23	K	54368	04062400	MICHIGAMME R NR WITCH LAKE MICH	461448	0880045	MI	043	STREAM	1964			*				GS
23	O	51205	04080000	LITTLE WOLF R AT ROYALTON WIS	442445	0885155	WI	135	STREAM	1963	1970		*				GS
23	O	56322	04081000	WAUPACA R NR WAUPACA WIS	441950	0885945	WI	135	STREAM	1963	1971		*				GS
23	Q	65484	04072750	LAWRENCE C NR WESTFIELD WIS	435352	0893452	WI	077	STREAM	1967			*	*	*		GS
23	Q	67735	04073500	FOX R AT BERLIN WIS	435715	0885710	WI	047	STREAM	1967			*				GS
23	R	67734	04085200	KEWAUNEE R NR KEWAUNEE WIS	442730	0873323	WI	061	STREAM	1968			5	8	8		GS
26	A	51206	04086000	SHEBOYGAN R AT SHEBOYGAN WIS	434425	0874535	WI	117	STREAM	1963			*				GS
26	B	51207	04087000	MILWAUKEE R AT MILWAUKEE WIS	430600	0875430	WI	079	STREAM	1963			5	8	8		GS
26	B	67732	04086200	E B MILWAUKEE R AT NEW FANE WIS	433301	0881118	WI	039	STREAM	1968			5	8	8		GS
26	B	67731	04086340	N B MILWAUKEE R NR FILLMORE WIS	432858	0880339	WI	131	STREAM	1968			5	8	8		GS
26	B	67730	04086360	MILWAUKEE R AT WAUBEKA WIS	432822	0875923	WI	089	STREAM	1968			*				GS

MAP NUMBER	MAP LETTER	OWDC NUMBER	AGENCY STATION NUMBER	STATION NAME	LATITUDE	LONGITUDE	STATE	COUNTY	SITE	PERIOD OF RECORD BEGAN	PERIOD OF RECORD DISCONTINUED	INTERRUPTED RECORD	SEDIMENT CONCENTRATION SUSPENDED	SEDIMENT PARTICLE SIZE SUSPENDED	SEDIMENT PARTICLE SIZE BED MATERIAL	SEDIMENT OTHER	AGENCY REPORTING
26	B	67718	40865	CEDAR R NR CEDARBURG WIS	431925	0875848	WI	089	STREAM	1968	1971		*				GS
26	C	73122	04087257	PIKE R NR RACINE WIS	423849	0875138	WI	059	STREAM	1971			5	8	8		GS
26	C	51209	04087240	ROOT R AT RACINE WIS	424505	0874925	WI	101	STREAM	1964			5	8	8		GS
26	C	51208	04087220	ROOT R NR FRANKLIN WIS	425225	0875945	WI	079	STREAM	1964			5	8	8		GS
27	A	51200	04025500	BOIS BRULE R AT BRULE WIS	463215	0913545	WI	031	STREAM	1964			8				GS
27	A	73123	04026005	BOIS BRULE R NR L SUPERIOR WIS	464220	0913607	WI	031	STREAM	1971			5	7	7		GS
27	A	67729	04026300	SIOUX R NR WASHBURN WIS	464120	0905702	WI	007	STREAM	1968			8				GS
27	A	57119	NE-8	NE NEMADJI R	463107	0922322	MN	017	STREAM	1967	1968					5	111
27	B	51201	04027000	BAD R NR ODANAH WIS	462915	0904145	WI	003	STREAM	1964			5				GS
29	A	54362	04001000	WASHINGTON C AT WINDIGO MICH	475523	0890842	MI	083	STREAM	1964			8		7		GS
29	A	63278	04014500	BAPTISM R NR BEAVER BAY MINN	472015	0911200	MN	075	STREAM	1962		*	*	*			GS
29	B	72871	SLB 0	SUPERIOR BAY LIFT BRIDGE			MN	137	STREAM	1971						5	111
29	D	63279	04018750	ST LOUIS R AT FORBES MINN	472148	0923556	MN	137	STREAM	1967	1971		5	7			GS
29	D	57120	SL-19.1	SAINT LOUIS R	463910	0921655	MN	137	STREAM	1953						5	111
29	D	57122	SL-120	SAINT LOUIS R	472147	0923550	MN	137	STREAM	1967	1968					5	111
29	D	57121	SL-51.9	SAINT LOUIS R	465050	0923434	MN	137	STREAM	1962						5	111
30	A	57123	LS-3	LAKE SUPERIOR	465154	0915730	MN	137	LAKE	1967	1968					5	111
30	A	57124	LS-2	LAKE SUPERIOR	471810	0911407	MN	075	STREAM	1967						5	111
30	A	57125	LS-1	LAKE SUPERIOR	474411	0902314	MN	031	LAKE	1967						5	111
30	B	55958	6373-496	LK MICHIGAN GREAT LAKES ILL	421847	0874809	IL	097	LAKE	1960			3				NFF

PART B – QUALITY OF SURFACE WATER

Stations in this Part are arranged in downstream order. That is, stations are listed in a downstream direction along the main stream, and stations on tributaries are listed between main-stream stations that are upstream and downstream from the mouth of the tributary. Stations on tributaries that enter above the first main-stream station are listed before the first main-stream station. Stations on tributaries to tributaries are listed in a similar manner.

Pages in this listing are arranged so that station identification, including period of record and storage of data, are on the left-hand pages, and the types of data available are on the facing right-hand pages. Following are brief explanations of the major column headings of the station listings in Part B:

Explanation of Major Column Headings in Part B

Map Number
Geographic unit and subunit in which station is located (see Fig. 1).

OWDC Number
This number, assigned by OWDC, is used to identify a specific data-acquisition activity included in the Catalog, and to crosstie the entry in the listing with the plot on the companion station-location maps.

Agency Station Number and Name
Assigned by reporting agency. To meet space limitations, the following abbreviations generally are used in the station names; in some instances, others may have been used:

AB	Above	MF	Middle Fork
BL	Below	NF	North Fork
B	Branch	SF	South Fork
EB	East Branch	WF	West Fork
MB	Middle Branch	L	Little
NB	North Branch	LK	Lake
SB	South Branch	M	Middle
WB	West Branch	N	North
BK	Brook	NR	Near
C	Creek	P	Pond
CA	Canal	R	River
CL	Coulee	RE	Reservoir
D	Ditch	RN	Run
DI	Distributary	S	South
E	East	STR	Stream
ES	Estuary	TR	Tributary
F	Fork	W	West
EF	East Fork		

Latitude-Longitude
As given by the reporting agency.

State and County
Symbols are used for States (and other areas), and code numbers are used for counties and independent cities in accordance with Federal Information Processing Standards (see Table 4).

Site
Type of water body for which data are acquired at the listed station: Stream, Canal, Lake, Reservoir (Reser), Estuary (Estary), Spring, Drain, Other.

Period of Record
Calendar year of beginning or ending of station activity.

Interrupted Record
Asterisk (*) indicates that during the period of record, data acquisition was suspended one or more times for an interval of one year or more.

Storage of Data
Asterisk (*) indicates in what form data are stored.

Types of Data
Frequency of determination or measurement of a parameter listed is indicated by the following numbers. If part or all of the data are telemetered, this is shown by the letter "T."

1. Continuous
2. Seasonal
3. Daily
4. Weekly
5. Monthly
6. Quarterly
7. Annual
8. Other periodic

An asterisk (*) indicates that the parameter has been measured, but the measurement was discontinued, or that frequency is unknown.

Supplementary data
Asterisks (*) are used in appropriate columns to indicate other types of water-data activities at the site.

Agency Reporting
Agency codes are given in Table 1.

III-4 Table 2 Reservoir Sediment Surveys in the U. S. Portion of the Great Lakes Basin[1/]

Reservoir	Nearest Town - State	Date of Survey	Net Drainage Area Sq Km	Net Drainage Area Sq Mi	Storage Capacity Cu M	Storage Capacity Ac Ft	Total Sediment Accum. Cu M	Total Sediment Accum. Ac Ft	Average Annual Sediment Accum. Cu M	Average Annual Sediment Accum. Ac Ft
1. Lake Rockwell	Kent, Ohio	Aug. 1914	321.4	(124.10)	9,160,000	(7,423.0)	-	-	-	-
ditto	ditto	Aug. 1950	ditto	ditto	8,499,000	(6,887.0)	661,400	(536.0)	18,400	(14.88)
2. Babb Pond	Richfield, Ohio	1932	0.05	(0.02)	302	(0.245)	-	-	-	-
ditto	ditto	Apr. 1951	ditto	ditto	233	(0.189)	69	(0.056)	4	(0.003)
3. Basom Pond	Hudson, Ohio	1944	0.83	(0.32)	4,780	(3.87)	-	-	-	-
ditto	ditto	Apr. 1951	ditto	ditto	4,000	(3.24)	780	(0.63)	120	(0.096)
4. Christener Pd.	Parma, Ohio	1940	0.23	(0.09)	4,200	(3.40)	-	-	-	-
ditto	ditto	Apr. 1951	ditto	ditto	3,440	(2.79)	750	(0.61)	68	(0.055)
5. Schoenbeck Pd.	Richfield, Ohio	1940	0.08	(0.03)	1,830	(1.48)	-	-	-	-
ditto	ditto	Apr. 1951	ditto	ditto	1,590	(1.29)	230	(0.19)	21	(0.017)
6. East Branch	Burton, Ohio	1939	43.72	(16.88)	5,749,000	(4,659.0)	-	-	-	-
ditto	ditto	Jun. 1949	ditto	ditto	5,596,000	(4,535.0)	153,000	(124.0)	15,300	(12.4)
7. Centerville Mills	Aurora, Ohio	1855	26.9	(10.38)	106,500	(86.3)	-	-	-	-
ditto	ditto	1949	ditto	ditto	47,300	(38.3)	54,300	(44.0)	580	(0.468)
8. Grand	Celina, Ohio	1844	240.9	(93.0)	160,636,000	(130,175)	-	-	-	-
ditto	ditto	Aug. 1940	ditto	ditto	131,551,000	(106,605)	29,085,000	(23,570.0)	303,000	(245.5)
9. Goller Pond	Defiance, Ohio	Mar. 1945	0.06	(0.024)	11,700	(9.5)	-	-	-	-
ditto	ditto	Aug. 1951	ditto	ditto	11,600	(9.4)	123	(0.1)	19	(0.015)
10. Auglaizer Power	Defiance, Ohio	1912	6,024	(2,326.0)	17,770,000	(14,400.0)	-	-	-	-
ditto	ditto	1951	ditto	ditto	14,314,000	(11,600.0)	3,455,200	(2,800.0)	88,500	(71.75)

1/ From Great Lakes Basin Framework Study, App. 18, Erosion and Sedimentation.

Table 2 continued

Reservoir	Nearest Town - State	Date of Survey	Net Drainage Area Sq Km	Net Drainage Area Sq Mi	Storage Capacity Cu M	Storage Capacity Ac Ft	Total Sediment Accum. Cu M	Total Sediment Accum. Ac Ft	Average Annual Sediment Accum. Cu M	Average Annual Sediment Accum. Ac Ft
11. Eagle Creek	Defiance, Ohio	1912	13.5	(5.2)	159,000	(129.0)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	91,300	(74.0)	67,870	(55.0)	1,740	(1.41)
12. Beetree Creek	Defiance, Ohio	1912	5.0	(1.91)	182,600	(148.0)	-	-	-	-
ditto	ditto	Aug. 1951	ditto	ditto	128,300	(104.0)	54,300	(44.0)	1,390	(1.13)
13. Batt Pond	Defiance, Ohio	Apr. 1947	0.03	(0.012)	3,200	(2.6)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	3,090	(2.5)	123	(0.1)	28	(0.023)
14. Harrison Lake	Fayette, Ohio	1941	96.0	(37.0)	1,223,000	(991.0)	-	-	-	-
ditto	ditto	Jun. 1949	ditto	ditto	1,146,500	(929.1)	76,400	(61.9)	9,220	(7.47)
ditto	ditto	Jul. 1951	ditto	ditto	1,113,500	(902.4)	33,000	(26.7)	15,700	(12.7)
15. Allmandinger Pd	Ohio City, Ohio	Jan. 1945	0.09	(0.035)	6,270	(5.08)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	5,810	(4.71)	457	(0.37)	68	(0.055)
16. Bucyrus #2	Bucyrus, Ohio	1919	7.2	(2.79)	298,600	(242.0)	-	-	-	-
ditto	ditto	Jun. 1949	ditto	ditto	269,000	(218.0)	29,620	(24.0)	988	(0.8)
17. Contris Pond	Lafayette	1947	0.34	(0.13)	11,350	(9.2)	-	-	-	-
ditto	ditto	1951	ditto	ditto	9,750	(7.9)	1,600	(1.3)	401	(0.325)
18. Sixmile Creek	Defiance, Ohio	1912	55.4	(21.4)	1,227,800	(995.0)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	858,900	(696.0)	369,000	(299.0)	9,470	(7.67)
19. Burt Lake	Oakwood, Ohio	Sep. 1948	1.9	(0.74)	72,800	(59.0)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	70,340	(57.0)	2,470	(2.0)	881	(0.714)
20. Kohart Pond	Grover Hill, Ohio	Sep. 1943	0.05	(0.019)	2,950	(2.4)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	2,590	(2.1)	123	(0.1)	16	(0.0128)

Table 2 continued

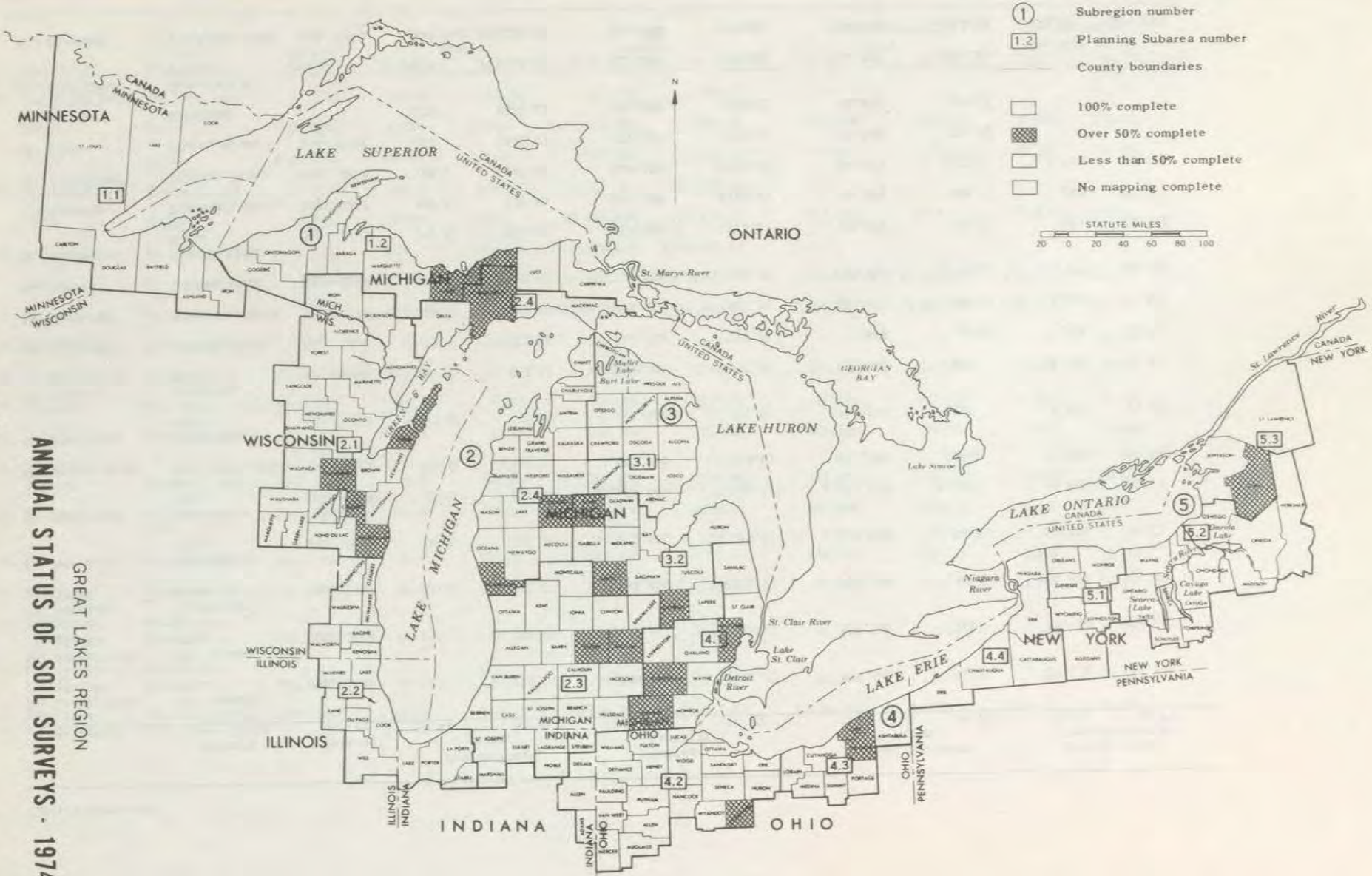
Reservoir	Nearest Town - State	Date of Survey	Net Drainage Area Sq Km	Net Drainage Area Sq Mi	Storage Capacity Cu M	Storage Capacity Ac Ft	Total Sediment Accum. Cu M	Total Sediment Accum. Ac Ft	Average Annual Sediment Accum. Cu M	Average Annual Sediment Accum. Ac Ft
21. Van Buren Lake	Findlay, Ohio	1939	58.8	(22.72)	306,000	(248.0)	-	-	-	-
ditto	ditto	Nov. 1948	ditto	ditto	253,000	(205.0)	53,100	(43.0)	5,580	(4.52)
ditto	ditto	Aug. 1951	ditto	ditto	229,500	(186.0)	23,400	(19.0)	8,370	(6.78)
22. Lake Rushford	Coneadea, N.Y.	1925	157.2	(60.7)	34,552,000	(28,000.0)	-	-	-	-
ditto	ditto	1951	ditto	ditto	33,844,000	(27,426.0)	708,300	(574.0)	27,270	(22.1)
23. Mount Morris	Mt. Morris, N.Y.	1951	2619	(1011.0)	417,104,000	(338,010.0)	-	-	-	-
ditto	Mt. Morris, N.Y.	1957	ditto	ditto	415,378,000	(336,611.0)	1,714,000	(1,389.0)	285,000	(231.0)
ditto	Mt. Morris, N.Y.	1963	ditto	ditto	413,875,000	(335,393.0)	3,106,000	(2,517.0)	269,000	(218.0)
24. Orchard Park	Buffalo, N.Y.		4.4	(1.7)	-	-	-	-	284	(0.23)
25. Saline Mill	Saline, Mich.	Mar. 1969	163.2	(63.0)	296,000	(240.1)	136,400	(110.5)	4,390	(3.56)
26. Bridgeway	Ann Arbor, Mich.	Mar. 1969	19.4	(7.5)	94,600	(76.7)	35,500	(28.8)	864	(0.70)
27. Franklin Mill	Franklin, Mich.	Apr. 1969	20.2	(7.8)	120,700	(97.8)	104,500	(84.7)	790	(0.64)
28. Tecumseh (Evans)	Tecumseh, Mich.	Apr. 1969	68.1	(26.3)	281,100	(227.8)	164,200	(133.1)	1,160	(0.94)
29. Sharon Hollow	Manchester, Mich.	May, 1969	64.8	(25.0)	318,500	(258.1)	140,900	(114.2)	3,330	(2.7)
30. Norvell	Norvell, Mich.	May, 1969	65.5	(25.3)	885,500	(717.6)	265,700	(215.3)	2,650	(2.15)
31. Brooklyn	Brooklyn, Mich.	May, 1969	16.1	(6.2)	307,600	(249.3)	77,700	(63)	3,700	(3.0)
32. Manchester (Power)	Manchester, Mich.	May, 1969	16.6	(6.4)	356,500	(288.9)	36,300	(29.4)	1,580	(1.28)
33. Manchester (Mill)	Manchester, Mich.	May, 1969	44.0	(17.0)	26,300	(21.3)	12,960	(10.5)	210	(0.17)
34. Kent Lake	Milford, Mich.	Jun. 1969	114.0	(44.0)	26,166,000	(21,204.0)	2,628,000	2,130)	136,600	(110.7)

Table 2 continued

Reservoir	Nearest Town - State	Date of Survey	Net Drainage Area		Storage Capacity		Total Sediment Accum.		Average Annual Sediment Accum.	
			Sq Km	Sq Mi	Cu M	Ac Ft	Cu M	Ac Ft	Cu M	Ac Ft
35. Stony Creek (North)	Mt. Vernon, Mich.	Jun. 1969	145.0	(56.0)	1,229,100	(996.0)	139,440	(113)	22,130	(17.93)
36. Stony Creek (South)	Mt. Vernon, Mich.	Jun. 1969	145.0	(56.0)	4,848,000	(3,929.0)	330,700	(268)	52,490	(42.54)
37. Oakwoods Metro.	Flat Rock, Mich.	Jun. 1969	81.8	(31.6)	1,161,400	(941.2)	372,000	(301.5)	8,450	(6.85)
38. Belleville	Belleville, Mich.	Jul. 1969	52.6	(20.3)	24,612,000	(19,945.0)	2,424,800	(1,965)	60,710	(49.2)
39. Ford Lake	Ypsilanti, Mich.	Jul. 1969	29.0	(11.2)	22,121,000	(17,926.0)	2,271,800	(1,841)	63,060	(51.1)
40. Barton Pond	Ann Arbor, Mich.	Jul. 1969	474.0	(183.0)	3,887,100	(3,150.0)	677,500	(549)	12,550	(10.17)
41. Iron Mill	Manchester, Mich.	Aug. 1969	13.5	(5.2)	1,913,900	(1,551.0)	485,000	(393)	4,850	(3.93)
42. Tecumseh Red (Mill)	Tecumseh, Mich.	Aug. 1969	67.1	(25.9)	835,400	(677.0)	420,800	(341)	4,210	(3.41)
43. H. N. Fry	Onsted, Mich.	Aug. 1969	32.4	(12.5)	149,700	(121.3)	6,540	(5.3)	938	(0.76)
44. Newburgh	Plymouth, Mich.	Sep. 1969	140.6	(54.3)	824,100	(667.8)	129,400	(104.9)	3,590	(2.91)
45. Adrian	Adrian, Mich.	Sep. 1969	152.8	(59.0)	1,234,000	(1,000.0)	183,900	(149.0)	6,570	(5.32)
46. Waterford	Northville, Mich.	Sep. 1969	140.0	(54.0)	213,480	(173.0)	88,800	(72.0)	888	(0.72)
47. Phoenix	Plymouth, Mich.	Sep. 1969	147.1	(56.8)	277,700	(225.0)	65,800	(53.3)	654	(0.53)
48. Fenton Mill	Fenton, Mich.	Jan. 1970	116.6	(45.0)	549,100	(445.0)	236,900	(192.0)	1,780	(1.44)
49. Elsie	Elsie, Mich.	Nov. 1964	497	(192.0)	137,000	(111.0)	72,700	(58.9)	-	-
50. Rockford	Rockford		583	(225.0)	109,800	(89.0)	55,000	(44.6)	-	-
51. Fish Creek	Carsonville, Mich.		319	(123.0)	122,200	(99.0)	36,200	(29.3)	-	-
52. Stronach	Wellston, Mich.	Jan. 1953	604	(233.0)	784,800	(640.0)	756,400	(613.0)	18,450	(14.95)

GREAT LAKES REGION

ANNUAL STATUS OF SOIL SURVEYS - 1974



Section III-5

MOBILITY OF SEDIMENT WITHIN SURFACE AND GROUND-WATER SYSTEMS

Erosion is caused by and sediment derived from the action of moving water, ice, or wind on rock and soil. Of the above agents moving water is predominant in the Great Lakes Region.

The transport of sediment occurs by two general processes, as a suspension in the water and as a bedload movement. When sediment is transported in suspension in the water it is referred to as "suspended load" and when it is moved along the bottom it is called "bedload." There are apparently no sharp divisions between the two forms of transport except that bedload moves close to the channel bottom whereas the suspended load is dispersed throughout the depth and width of the moving water. The physical mechanism of both forms of transport is very complex and it is beyond the purview of this report to explore its nature.

Generally, bedload movement involves the more coarse textured, heavy materials that require relatively high channel velocities. Suspended sediment movement occurs in moving water over a much wider range of velocities. In both forms of sediment transport, depth of flow, velocity of flow, and the nature of the material being transported are important factors in sediment transport rates.

The quantitative role that each of two broad forms play in the transport of sediment in the Great Lakes Basin is not known. Much evidence, visual observation, and logic leads to the conclusion that the majority of sediment transport occurs as suspended sediment. It is estimated that the quantity of bedload will not exceed 10 percent of the suspended load.

Sediment does not often get into the ground-water system. This is because the soil acts as a very effective filter and removes the sediment very soon after water begins to move through the soil. It is conceivable sediment laden water could get into the ground-water through fractures in rock or by overflowing into wells which penetrate fractured rock. It is not felt that this condition is prevalent.

Section III-6

NATURAL RENOVATION MECHANISM AVAILABLE TO REMOVE POLLUTANTS AND CONTAMINANTS (PHYSICAL, CHEMICAL, BIOLOGICAL)

Natural renovation mechanisms that remove sediment which are different from the natural process now utilized in soil conservation practices are limited. The most common and effective mechanism used is vegetation. Given sufficient plant cover soil erosion can be held to a minimum. Many of the soil conservation practices listed in Section V are based on this principle.

The effectiveness of vegetation as an erosion retardant varies widely. For example, erosion rates on a field can be reduced by 60% by converting from continuous corn to a rotation of two years corn, one year small grain, and one year grass and clover meadow. Conversion to continuous pasture would reduce erosion even more while conversion from close growing crops to pasture would reduce erosion at a lesser rate. Another example might be exclusion or management of livestock to allow the natural plant growth to regain adequate cover to protect the soil.

The other soil conservation practices listed in Section V also were developed to take advantage of natural processes. Debris basins and reservoirs are constructed to exploit the fact that gravity causes particles in suspension in the water to settle in the reservoir. Terraces and diversions reduce slope lengths. Minimum tillage, chiseling, incorporating residues into the soil and other such practices increase infiltration and decrease runoff. Other management, land treatment and structural measures utilize some natural physical process to effect a reduction in erosion or sediment yield. The effectiveness of these practices also varies widely. A reservoir can be from 100% effective to nearly ineffective depending on the reservoir trap efficiency. The other practices vary almost as much in effectiveness.

Both chemical and biological actions can take place which cause either retention of soils in place or deposition of sediment enroute. Vegetative growth which protects soils can be stimulated by fertilizers. Better soil structure which promotes infilgration and reduced runoff can be developed by the incorporation of crop residues into the soil. This interaction of the physical, chemical, and biological actions is extremely complex and in many cases more than one process is involved in reducing erosion and sediment yield. More often than not chemical and biological actions complement physical actions in the incorporation of conservation practices on the land.

Section III-7

PROBABLE CHANGES IN SEDIMENT YIELD IN TIME

For this section it was thought that the predicted sediment yields to the Great Lakes for specified years from agricultural and urban land would be informative. Data of this type were not available so it was developed in the following manner:

Data were available on the sediment yield from sheet erosion to each one of the Great Lakes (8). Since this figure included both forest and agricultural land the contribution from forest land needed to be deleted to leave only sheet erosion from agricultural land. This was done by applying a weighted factor which consisted of a combination of the erosion rates for each land use and amount of each land use. This factor times the sheet erosion figure gave the 1970 sediment yield from sheet erosion on agricultural land. Then a factor of 3 percent for gully erosion was added to the sheet erosion sediment yield figure to obtain the sediment yield from agricultural land from sheet and gully erosion for 1970.

To establish the sediment yields for future dates, the present sediment yield was multiplied by the percent change reflected in the Gross Erosion for Projected Dates (5). This table takes into account increased or decreased demand for different types of crops, projected advances in technology, displacement of agricultural land by urban growth, and the effect of an increased amount of land treatment. These projected changes are based on material regarding population and food and fiber requirements furnished by the Office of Business Economics (now the Bureau of Economic Analysis) - Economics Research Service and used in the Great Lakes Basin Framework Study (5).

The 1973 figures were obtained by interpolation of the 1970 and 1980 figures.

Table 3 through 7 of this section show Estimated Annual Sediment Yield from Sheet and Gully Erosion on Agricultural Land for the years 1976, 1980, 2000, and 2020.

The figures for years 1980, 2000, 2020 on Table 8, Estimated Average Annual Sediment Yield from Sheet and Gully Erosion on Urban Land to the Great Lakes, were determined by calculating a sediment delivery rate for each metropolitan area and multiplying it by the estimated erosion rate. The erosion rates were based on projected urban growth. The figure for 1973 was obtained from a direct proportion of the 1970 and 1980 erosion rate time the delivery rate for each metropolitan area.

TABLE 3

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE SUPERIOR [1/] [2/]

(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Superior Slope Complex	9.5	(10.5)	9.8	(10.7)	9.2	(10.1)	9.2	(10.1)
St. Louis River	2.5	(2.8)	2.6	(2.9)	2.4	(2.7)	2.4	(2.7)
Nemadji River	1.6	(1.8)	1.7	(1.9)	1.5	(1.7)	1.5	(1.7)
Apostle Complex	10.2	(11.2)	10.3	(11.4)	9.8	(10.8)	9.8	(10.8)
Bad River	3.4	(3.7)	3.4	(3.8)	3.3	(3.6)	3.3	(3.6)
Montreal River	1.0	(1.1)	1.0	(1.1)	1.0	(1.1)	1.0	(1.1)
Porcupine Mt. Complex	3.2	(3.5)	3.1	(3.4)	3.0	(3.3)	3.0	(3.3)
Ontonagon River	0.8	(0.9)	0.8	(0.9)	0.7	(0.8)	0.7	(0.8)
Keweenaw Pen. Complex	4.2	(4.6)	4.1	(4.5)	3.9	(4.3)	3.9	(4.3)
Sturgeon River	0.6	(0.7)	0.6	(0.7)	0.5	(0.6)	0.5	(0.6)
Huron Mt. Complex	1.8	(2.0)	1.7	(1.9)	1.6	(1.8)	1.6	(1.8)
Grand Marais Complex	2.2	(2.4)	2.1	(2.3)	2.0	(2.2)	2.0	(2.2)
Tahquamenon River	0.4	(0.4)	0.4	(0.4)	0.4	(0.4)	0.4	(0.4)
Sault Complex	0.5	(0.6)	0.5	(0.6)	0.5	(0.5)	0.5	(0.5)
TOTAL	41.9	(46.2)	42.1	(46.5)	39.8	(43.9)	39.8	(43.9)

[1/] All figures rounded to nearest 100 tons

[2/] Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 4

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE MICHIGAN 1/ 2/

(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Seul Choix Groscap	1.9	(2.1)	1.8	(2.0)	1.5	(1.7)	1.8	(2.0)
Manistique River	0.6	(0.7)	0.6	(0.7)	0.5	(0.6)	0.6	(0.7)
Bay De Noc	6.7	(7.4)	6.4	(7.0)	5.5	(6.1)	6.6	(7.3)
Escanaba River	4.4	(4.8)	4.2	(4.6)	3.6	(4.0)	4.4	(4.8)
Menominee Complex	4.4	(4.9)	4.3	(4.7)	3.7	(4.1)	4.4	(4.9)
Menominee River	19.9	(21.9)	19.0	(20.9)	16.6	(18.3)	19.7	(21.7)
Peshtigo River	4.2	(4.6)	4.0	(4.4)	3.5	(3.9)	4.2	(4.6)
Oconto River	26.2	(28.9)	25.0	(27.6)	22.0	(24.0)	26.0	(28.7)
Saumico Complex	33.9	(37.4)	32.3	(35.6)	28.3	(31.2)	33.7	(37.1)
Fox River	80.9	(89.2)	77.2	(85.1)	67.7	(74.6)	80.5	(88.7)
Green Bay Complex	127.8	(140.9)	121.9	(134.4)	106.8	(117.7)	127.0	(140.0)
Chicago-Milwaukee Complex	284.9	(314.1)	263.6	(290.6)	220.6	(243.2)	187.3	(206.5)
St. Joseph River	207.5	(228.7)	207.8	(229.1)	204.7	(225.6)	248.5	(273.9)
Kalamazoo River	187.5	(206.7)	187.8	(207.0)	185.0	(203.9)	224.5	(247.5)
Grand River	94.6	(104.3)	81.7	(90.1)	68.5	(75.5)	84.4	(93.0)
Black River	25.7	(28.3)	22.1	(24.4)	18.6	(20.5)	22.9	(25.2)
Muskegon River	46.1	(50.8)	39.8	(43.9)	33.4	(36.8)	41.1	(45.3)
Sable Complex	9.4	(10.8)	8.2	(9.0)	6.8	(7.5)	8.4	(9.3)
Manistee River	20.0	(22.0)	17.2	(19.0)	14.4	(15.9)	17.8	(19.6)
Traverse Complex	70.6	(77.8)	61.0	(67.2)	51.2	(56.4)	63.0	(69.4)
TOTAL	1,257.2	(1,385.9)	1,185.9	(1,307.3)	1,062.9	(1,171.7)	1,206.8	(1,330.2)

1/ All figures rounded to nearest 100 tons

2/ Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 5

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE HURON [1/] [2/]

(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Cheboygan River	1.4	(1.5)	1.4	(1.5)	1.2	(1.3)	1.2	(1.3)
Presque Isle Complex	2.4	(2.7)	2.4	(2.6)	2.1	(2.3)	2.1	(2.3)
Thunder Bay River	5.0	(5.5)	4.9	(5.4)	4.3	(4.7)	4.3	(4.7)
Alcona Complex	1.6	(1.8)	1.6	(1.8)	1.4	(1.5)	1.4	(1.5)
AuSable River	3.3	(3.6)	3.2	(3.5)	2.8	(3.1)	2.8	(3.1)
Rifle-AuGres Complex	12.9	(14.2)	12.7	(14.0)	11.2	(12.3)	11.2	(12.3)
Kawkawlin Complex	16.2	(17.9)	16.4	(18.1)	10.0	(11.0)	12.3	(13.6)
Saginaw River	115.2	(12.7)	116.6	(128.5)	70.9	(78.1)	87.5	(96.4)
Thumb Complex	60.2	(66.4)	61.0	(67.2)	37.0	(40.8)	45.7	(50.4)
TOTAL	218.2	(240.6)	220.2	(242.6)	140.9	(155.1)	168.5	(185.6)

[1/] All figures rounded to nearest 100 tons

[2/] Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 6

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE ERIE 1/ 2/
(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Black River	29.3	(32.3)	28.3	(31.2)	24.8	(27.3)	24.6	(27.1)
St. Clair Complex	25.1	(27.7)	24.2	(26.7)	21.1	(23.3)	21.0	(23.2)
Clinton River	41.7	(46.0)	40.3	(44.4)	35.2	(38.8)	34.9	(38.5)
Rouge Complex	33.8	(37.3)	32.7	(36.0)	28.5	(31.4)	28.3	(31.2)
Huron River	57.1	(62.9)	55.1	(60.7)	48.1	(53.0)	47.7	(52.6)
Raisin River	91.5	(100.9)	88.4	(97.4)	77.2	(85.1)	76.7	(84.5)
Maumee River	1,092.8	(1,204.6)	1,120.5	(1,235.1)	1,157.7	(1,276.1)	1,215.7	(1,340.1)
Toussaint-Portage Complex	108.6	(119.7)	111.3	(122.7)	115.0	(126.8)	120.8	(133.2)
Sandusky River	109.7	(120.9)	215.5	(237.6)	222.7	(245.5)	233.9	(257.8)
Huron-Vermillian Complex	138.7	(152.9)	142.2	(156.7)	146.9	(161.9)	154.3	(170.1)
Black-Rocky Complex	89.4	(98.5)	80.1	(88.3)	81.8	(90.2)	89.1	(98.2)
Cuyahoga River	86.5	(95.3)	77.5	(85.4)	79.2	(87.3)	86.2	(95.0)
Grand River	179.8	(198.2)	161.1	(177.6)	164.7	(181.5)	179.2	(197.5)
Ashtabula-Conneaut Complex	11.3	(12.5)	10.2	(11.2)	10.3	(11.4)	11.2	(12.4)
Erie-Chautauqua Complex	7.1	(7.8)	6.5	(7.2)	5.0	(5.5)	5.0	(5.5)
Cattaraugus River	14.1	(15.5)	13.1	(14.4)	9.9	(10.9)	10.0	(11.0)
Tonawanda Complex	23.0	(25.3)	21.3	(23.5)	16.1	(17.8)	16.2	(17.9)
TOTAL	2,139.5	(2,358.3)	2,228.3	(2,456.1)	2,244.2	(2,473.8)	2,354.8	(2,595.8)

1/ All figures rounded to nearest 100 tons

2/ Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 7

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE ONTARIO [1/] [2/]

(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Niagara-Orleans Complex	48.7	(53.7)	43.9	(48.4)	41.6	(45.9)	41.6	(45.9)
Genesee River	59.7	(65.8)	53.9	(59.4)	51.1	(56.3)	51.1	(56.3)
Wayne-Cayuga Complex	49.5	(54.6)	44.4	(48.9)	31.2	(34.4)	36.6	(40.3)
Oswego River	99.4	(109.6)	89.2	(98.3)	62.7	(69.1)	73.4	(80.9)
Salmon-Perch Complex	43.0	(47.4)	38.6	(42.5)	27.0	(29.9)	31.8	(35.0)
Black River	24.5	(27.0)	22.0	(24.3)	20.7	(22.8)	20.7	(22.8)
Oswegatchie River	17.5	(19.3)	15.8	(17.4)	14.8	(16.3)	14.8	(16.3)
Grass-Raquette-St. Regis Complex	20.5	(22.6)	18.5	(20.4)	17.3	(19.1)	17.3	(19.1)
TOTAL	362.8	(400.0)	326.3	(359.6)	266.4	(293.8)	287.3	(316.6)

[1/] All figures rounded to nearest 100 tons

[2/] Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 8

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON URBAN LAND (U.S. PORTION) TO THE GREAT LAKES 1/ 2/

(TONS X 1000)

METROPOLITAN COMPLEX	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Lake Superior								
Duluth-Superior	13.8	(15.2)	14.2	(15.6)	16.0	(17.6)	18.0	(19.8)
SUB TOTAL	13.8	(15.2)	14.2	(15.6)	16.0	(17.6)	18.0	(19.8)
Lake Michigan								
Green Bay-Oskash	1.7	(1.9)	1.8	(2.0)	2.3	(2.5)	2.9	(3.2)
Milwaukee	3.6	(4.0)	4.0	(4.4)	5.4	(5.9)	7.1	(7.8)
Chicago	28.7	(31.6)	30.9	(34.1)	38.1	(42.0)	47.0	(51.8)
South Bend-Elkhart	5.3	(5.8)	5.5	(6.1)	6.7	(7.4)	8.3	(9.1)
Kalamazoo-Battle Creek	7.9	(8.7)	8.8	(9.7)	12.0	(13.2)	15.9	(17.5)
Grand Rapids	2.6	(2.9)	2.9	(3.2)	3.6	(4.0)	4.5	(5.0)
Lansing-Jackson	2.7	(3.0)	3.0	(3.3)	4.1	(4.5)	5.4	(6.0)
SUB TOTAL	52.5	(57.9)	56.9	(62.8)	72.2	(79.5)	91.1	(100.4)
Lake Huron								
Bay City-Saginaw-Flint	5.4	(5.9)	5.8	(6.4)	8.3	(9.1)	9.4	(10.4)
SUB TOTAL	5.4	(5.9)	5.8	(6.4)	8.3	(9.1)	9.4	(10.4)
Lake Erie								
Detroit	50.5	(55.7)	55.7	(61.4)	71.1	(78.4)	91.4	(100.8)
Toledo	5.1	(5.6)	5.8	(6.4)	6.7	(7.4)	8.4	(9.3)
Fort Wayne	6.5	(7.2)	7.3	(8.0)	10.0	(11.0)	13.6	(15.0)
Loraine-Elyria	11.3	(12.4)	12.1	(13.3)	15.1	(16.6)	18.7	(20.6)
Cleveland-Akron	102.4	(112.9)	109.4	(120.6)	136.5	(150.5)	170.0	(187.4)
Erie	9.2	(10.1)	9.7	(10.7)	11.5	(12.8)	14.3	(15.6)
SUB TOTAL	185.0	(203.9)	200.0	(220.4)	250.9	(276.7)	316.4	(348.7)

From the mid-1800s to about 1953, the Lake Superior region produced the bulk of the iron ore used in the nation's steel industry. In 1953, the peak year, production reached 97 million metric tons; by 1968, it was 57.5 million. The decline can be traced to three factors: depletion of high-grade reserves, the "direct-shipping" ores; imports of foreign ores; and the technological revolution that allowed production of highly desirable concentrates from previously unusable low-grade ores (taconite). Future production will be of taconite and similar ores on a very large scale, using production-line techniques. It is expected that about one-half of total U.S. needs for iron ore will be met by Lake Superior production; of this amount, Minnesota will produce 75 to 80 percent.

By 1880, Upper Michigan was the nation's leading producer of copper. Output declined steadily through 1954, except for temporary increases in the late 1920s and World War II. A new mine in Ontonagon County doubled the state's copper output, and by 1962 it was 74,000 tons, or 6.0 percent of national production. Activity at this mine remains strong, but the future of the world copper market is beset with uncertainty.

The history of iron and copper mining in the Lake Superior region is summarized by Deshpande and Dworsky (1973, p. 2-6 - 2-13).

3.3 Pollution Resulting from the Extractive Industries

3.3.1 Nonmetallic Extractive Industries Other Than Salt and Brines

In the average sand and gravel operation, pit-run material is washed, screened, and stockpiled in several sizes, from pebbles 3 inches or more in diameter down to fine sand. The effluent from washing may contain unwanted sand (grains larger than 1/16 mm in diameter) or silt (1/16-1/256 mm), but is seldom burdened with clay (finer than 1/256 mm). Little clay occurs with sand and gravel, because the water currents that originally deposited these materials -- meltwater from the ice sheets of the Pleistocene age -- were too swift to allow clay-size material to settle out. The effluent can be readily cleared of its sand or silt by settling in a pond, and this is commonly done, so that the water can be used over again. (People in the sand and gravel business say, "We use the water till we wear it out.") Such waters carry little dissolved material, other than what they originally had, as the constituents of sand and gravel are by their very nature resistant to chemical attack and thus unlikely to yield a solution load to waters that come into contact with them. Nearly all the waters used at sand and gravel pits are surface waters. At some deposits, the desired material lies below the water table, and is extracted from under water by dragline or clamshell bucket. The bodies of water that are left in worked-out pits constitute park lakes, waterways for residential development, or ground-water recharge basins. Pre-planning toward these goals is standard procedure at many of the larger pits. Constraints on the industry are imposed chiefly by (1) competition with urban expansion for close-in sites of production; (2) community objections to noise, traffic, and unsightliness; and (3) necessity for reclamation of mined-out areas.

The following paragraphs are quoted from a letter of March 14, 1974, from E. K. Davison, Director of Environmental Affairs of the National Sand and Gravel Association. In cooperation with a man from the EPA, Mr. Davison "solicited information on discharges from a number of our member companies around the country."

> I found in my limited survey that in about one-fourth of those operations which discharge into a public watercourse the discharge is simply a pump-down to keep a pit dry or, more usually, to evacuate excess water, rather than a discharge of wash-water from the screening plant. Such pumps appear to be well removed from portions of pits used for receiving screening plant washwater.
>
> The few analyses of pump-down discharges that I received indicated that suspended solids contents exceeding about 12 to 15 mg/l would be unusual. Analyses furnished me for discharges involving settling ponds for process water (as distinguished from pump-down discharges) showed a median suspended solids content of 139 mg/l. These were mostly "one-shot" samples taken for filling out the permit applications under the Refuse Act Program. One of the EPA Regional Offices in a study of both sand and gravel and crushed stone operations found a median of 55 mg/l for suspended solids. This particular report corroborated the observation of both the EPA Effluent Guidelines man and this Association office that volumes of discharges related to tons of production vary so widely that only the pollutant concentration method, as distinguished from loading per unit of production, is practical. Suspended solids and pH are the usual parameters of concern. The pH is no problem in discharges from sand and gravel operations.
>
> I have never heard of any concern or contamination of underground water provided the pits or ponds are receiving only process water from sand and gravel handling.

Pollution of waters by the sand and gravel industry may be appreciable, but if so it is largely unrecognized. In the sources consulted for this report, we have found hardly any mention of this subject. It seems to have given rise to little if any regulatory legislation.

Water may enter into the crushed-stone operation if the stone is naturally muddy or shaly, but this is an undesirable condition and at most quarries the quality of the stone is such that washing is not necessary. Water may be clarified by settling in ponds, but it is probable that some mud and silt goes into adjacent waters. At a limestone quarry on Kelleys Island, Ohio, for example, silty water was discharged for some years into Lake Erie, where it produced a little delta. Slight chemical effect is generally made on waters that come into contact with limestone or dolomite for a few minutes during washing.

Production of clay consists simply of extraction from a pit and removal to the plant. The industry is relatively small in the Great Lakes area and little effect on quality of waters is to be expected. Much the same is true of peat. This substance is dug up, shredded or otherwise dry-processed, packed, and shipped.

Between 16.3 and 19.7 million liters (4.3 and 5.2 million gallons) of water are pumped every day from a gypsum mine on the Port Clinton peninsula on the south side of Lake Erie (L. Purcell, U.S. Gypsum Co., personal communication). This exceptionally large quantity of water is explained by the presence of fractures through which lake water is entering the mine. (Variation in the above figures reflects water level in the lake, which in turn depends on prevailing winds.) In May, 1974, the water had a pH of 7.72, an average sulfate content of about 1,380 mg/l, and total dissolved solids averaging about 2,400 mg/l (R. Sturtz, Ohio EPA, personal communication). Some of the water is used in the sink-float process of upgrading the gypsum in the mill; the rest is discharged into Sandusky Bay, an arm of Lake Erie. The state EPA does not consider this discharge as a serious source of pollution, owing to the great dilution that it immediately undergoes. However, the Ohio EPA does not have standards that apply to lakes and bays, only to streams. It is not possible to generalize from one gypsum mine to others, as the amount and nature of water will vary with the local conditions. Water pumped from a gypsum mine in western New York, for an example, is pure enough to be diverted, by contract with the state, into a stream at a guaranteed rate of 5,677 liters (1500 gallons) per minute (R. Runvik, U.S. Gypsum Co., personal communication).

> Coal was at one time mined by underground methods in Bay, Genesee, Huron, Midland, Saginaw, and Tuscola counties [Michigan] from the Saginaw Formation of the Pennsylvanian System. Coal production ceased due to the thinness of the coal beds, the low grade and high sulphur content of the coal, and the high cost of underground mining. No future coal production is anticipated (U.S. Bureau of Mines, 1971, p. 5-127).

Effects, if any, of the discontinued mines on the quality of waters is not known. Deleterious effects of exploratory drill holes are discussed in Section 3.5.

3.3.2 Extraction of Salt and Brines

Unlike sand, gravel, and the other relatively inert substances discussed in the preceding section, salt is readily dissolved in the earth's waters, and the resulting brines are chemically active. When discharged on the ground surface, they sterilize the soil and kill vegetation; if discharged into a water supply, they give an undesirable salty taste to drinking water. Thus "chloride contamination" is a serious and continuing problem in brine fields, and also at salt mines where mine seepage must be disposed of. Waste brines have traditionally been flushed into streams or lakes. For example, brine has been discharged into the south end of Seneca Lake, New York, for nearly 100 years; in addition, operational leakages, spillages, and the former practice of stockpiling salt have undoubtedly contributed chlorides to the lake.

> Seneca Lake had an average chloride content of about 100 mg/l from about 1920, when first analyzed, to 1960. From 1960 to 1970, chlorides increased to about 200 mg/l. Since 1970 there has been a levelling off or slight decline in chloride. Salt-plant operations (possibly including leaky abandoned salt wells) are suspected causes (R. M. Waller, U.S. Geol. Survey, personal communication).

Newly established standards controlling stream pollution in New York, discussed in Section 3.9.3, will eliminate disposal of brines into streams in that state.

Waste brines may be pumped into ponds, in the hope that the water fraction will evaporate and the salts remain as precipitates. This practice, however, is successful only in arid or semi-arid regions, where evaporation exceeds rainfall (U.S. EPA Rept. EPA-430/9-73-011, p. 94). Under the Great Lakes climate, evaporation is slow and much of the brine infiltrates into the ground-water supply. Leakage and filtration through the dikes at evaporation ponds is also a hazard. A third method of disposal is to pump the brine back into the ground. It may be pumped into an oil-bearing zone, where it may help to repressure the reservoir and act as a water-flood to move more oil toward well bores; needless to say, this is a technique that requires considerable specialized knowledge. Or, the brine may be pumped into a deep porous and permeable bed or rock that will absorb and hold it. Much remains to be learned about this technique.

In sum, the production of salt and brines appears to be the most damaging extractive industry to the quality of the water supply in the Great Lakes region.

3.3.3 Mining of Iron and Copper Ores

Crushing, screening, washing, and gravity beneficiation are the chief processes that iron ore goes through at the mine. The wash water, much of which used to run off and produce "red water" in the streams draining the Mesabi and other ranges, is now clarified in settling ponds and is re-used. The development of closed-water systems in the tailings-disposal areas has been notably successful in reducing pollution (R. K. Hogberg, Minnesota Geological Survey, personal communication). The St. Louis River at Duluth shows the lowest longtime average annual sediment yield of any large river system in the Great Lakes Basin: 1 metric ton per km^2. (The Cuyahoga River south of Cleveland shows 88 metric tons per km^2.) The St. Louis River drains a large part of the Minnesota iron ranges. Planning subareas 1.1 and 1.2 of the GLBC, which contain the iron and copper mines, have the lowest gross erosion rate of all subareas in the Basin, as indicated in Table 2. The effects of mining are not mentioned in a study of suspended sediment in streams tributary to Lake Superior (Callahan, 1973).

3.4 Assessing and Quantifying the Problems

To assess the extent of pollution by the region's extractive industries other than salt and brines would involve visiting and inspecting 1,000 to 2,000 separate pits, quarries, and mines. Most of the pollution would be in the form of sediment in water. Quantifying the observations would involve measuring the discharge and determining the concentration of suspended solids at each occurrence. To be of value for purposes of comparison and totalling, such measurements should be taken at times of average or "normal" discharge and pollution. It is doubtful whether the results would be reliable and significant enough to warrant the effort and expense.

Determining a water's chloride content is a relatively simple process of titration and is done on a routine basis. It gives quantitative results that are comparable with a large body of similarly expressed data from waters at many localities. Standards of comparison are generally known and accepted. For example, the U.S. Public Health Service recommends that drinking water contain no more than 250 mg/l of chloride. Thus chloride contamination may be readily assessed and quantified.

TABLE 2 - GROSS EROSION RATE, SUBAREAS 1.1 AND 1.2

Subarea	Tons per Acre per Year Current	1980	2020
1.1	0.33	0.34	0.32
1.2	0.24	0.23	0.22

Source: Great Lakes Basin Framework Study, Appendix 18, Table 4, 1970.

3.5 Mobility of Pollutants

The mobility of sediment discharged into water bodies, as from washing plants at gravel or crushed-stone operations, depends on (1) the coarseness of the material, and (2) the kinetic energy, if any, of the water body. Sand and silt discharged into a pond or other standing water will settle promptly. Clay may remain in suspension for a long time. If the discharge is into a stream, natural sorting will take place: coarse sand will settle nearest the point of inflow, and silt farther away; clay will very likely not come to rest until the stream reaches a body of quiet water where energy is at a minimum and settling can take place. Sources of severe pollution by suspended solids should be relatively easy to spot because of the above-described method of dispersal.

Brines are highly mobile. Those brines that seep into streams or lakes may cause contamination, or they may undergo such great dilution by fresh waters that the stream or lake waters show only slight effects. The chloride concentration of streams during low flow throughout Michigan rarely exceeds 35 mg/l, even though brines are present in the shallow subsurface of much of the state.

Waters that remain underground can hardly purge themselves. If fresh waters become contaminated with chloride, they may remain contaminated indefinitely. Under natural conditions, underground waters generally move very slowly; but when man enters the picture he may upset the hydrostatic equilibrium, causing rapid migration of brines into fresh-water zones. From 1860 on, hundreds of wells were drilled in the Saginaw Valley of Michigan, in the search for coal and valuable brines. Few if any of these wells were lined with casing, with the result that they served as conduits through which highly mineralized brines from deep in the rocks rose and mixed with near-surface fresh waters. Deterioration of ground-water quality was noticed at Saginaw as early as 1906 (Bowman, 1906); it was attributed to abandoned brine wells. At Lowell, near Grand Rapids, Kent County, the chloride content in a municipal well increased from a trace in 1933 to 925 mg/l in 1941. The major cause was leakage of brine through abandoned oil and gas wells (Deutsch, 1963). A well drilled in Lansing in 1956 contained less than 100 mg/l, but after two months of operation the chloride content had risen to about 900 mg/l. The probable source of contamination was an abandoned brine well (4,575 mg/l of sodium chloride), which had been drilled in 1867. Smith (1944) has described the deleterious effects of brine migration on overlying fresh-water aquifers in Michigan, and Smith and Frye (1945) have reported on the history of brine contamination in the Saginaw Valley.

Many thousands of shallow shot-holes are drilled each year in Michigan by oil companies for geophysical testing; borings are made for foundation work; and holes are drilled by cement companies or others to test the limestone bedrock. All these holes are uncased, and many have acted as pipes through which mobile brines move upward. The result has been chloride contamination of water supplies that continues even today.

The gross areal extent of ground-water contamination in southern Michigan is not known. Probably most individual areas do not exceed a few acres. In districts where ground water is pumped, the resulting decrease in hydrostatic head might lead to more widespread contamination, through open holes and inter-aquifer leakage.

In considerable areas of Wood and Hancock Counties, northwestern Ohio, surface and ground waters have a higher-than-normal salinity. Near Portage, Wood County, the Portage River at periods of low flow commonly contains more than 150 mg/l of chloride. It seems very likely that contamination is the result of leakage of brines from the old Lima oil and gas field. This field was discovered in 1884, reached its peak of production in 1896, and is nearly exhausted today. According to Orton (1888), oil seeps into wells and excavations in the Findlay area antedated discovery by 50 years. No doubt deep brines have been moving upward for scores of years, in large part via old oil and gas wells which were never cased or in which the casing has long since rusted away.

Another situation is exemplified by the artificial-brine industry near Syracuse, New York. The wells are at Tully, 27.3 km (17 miles) to the south; the brine is pumped to a plant near the west shore of Onondaga Lake, where caustic soda and soda ash are produced. Lagoon effluent and leachate from stock piles contribute high chloride content to Ninemile Creek and Onondage Lake. The situation is discussed in two reports (Kantrowitz, 1970; Shampine, 1973).

3.6 Natural Renovative Mechanisms

Water that is contaminated only by suspended solids, as at a gravel or crushed-stone plant or taconite mine, undergoes renovation if these solids are allowed to settle out. Water thus clarified may be re-used in the plant or discharged into water bodies without deleterious effects.

There is essentially only one natural mechanism for renovation of chloride-contaminated waters. This is dilution by fresh waters. Unless the pollution load is very heavy and the water body is small, such dilution ordinarily reduces the chloride level to acceptable values. Where underground waters have been invaded by brines, there is no natural means of renovation.

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3.7 Probable Changes in Extractive Industries with Time

It will be seen from Table 3 that all the commodities discussed in this report are expected to increase markedly in production well into the next century. Salt, the major source of water pollution, was first produced for food seasoning and preservative, but the salt industry has long since become a mainstay of the chemical complex, and the bulk of today's production goes into the sodium and the chlorine families of basic chemicals. This being the case, there is little doubt that demand for salt will continue to rise. It is expected to more than quintuple by the year 2020.

Production of both iron and copper is expected to increase moderately in the coming decades. Deleterious effects on water quality may increase proportionately, although it seems entirely within reason to believe they might be held to present levels or even made to decrease, as more is learned and applied to control of tailings accumulations and the drainage from them.

3.8 Seriousness of Future Pollution Problems

For mineral resources other than salt and brines, there is little basis for prediction. A reasonable assumption seems to be that pollution will increase in roughly the same proportion as production increases. We believe the over-all effects will continue to be slight.

As for salt and brines, it is hard to avoid the conclusion that chloride contamination will increase markedly unless firm governmental controls on extractive and manufacturing processes are imposed. Hazards to be expected are exemplified at a New York site, where a company started mining rock salt about 2 years ago. While working on a deep-well disposal system, they have stored mine seepage and plant wastes in holding ponds. There has been leakage, dike failure, and production of a fine saline dust, all of which have caused problems to the water supply (R. M. Waller, U.S. Gel. Survey, personal communication). With five times as much salt being produced in 2020 as is being produced today, it is clear that drastic controls and monitoring systems will be necessary.

3.9 Review of Programs of Controlling Pollution

A comprehensive description of methods for controlling pollution from extractive operations of all types is given in two reports of the U.S. Environmental Protection Agency (1973).

TABLE 3 - PROJECTED MINERAL PRODUCTION, GREAT LAKES REGION,
BY SELECTED COMMODITIES
(Thousands of Metric Tons)

Commodity	1968[1/]	1980	2020
Sand and gravel...	116,981	155,276	464,876
Crushed stone.....	100,298	130,151	387,537
Iron ore.....	57,548	66,605	126,748
Salt.....	W[2/]	20,993	94,911
Clay.....	3,755	4,600	11,663
Gypsum.....	W	2,005	3,692
Peat.....	237	262	427
Copper ore.....	68	91	299

[1/] Actual.

[2/] Withheld to avoid disclosing individual company data.

Source: Great Lakes Basin Framework Study, Table 5-103 (1971).

3.9.1 Michigan

Michigan's Mine Reclamation Act (Act 92, 1970, amended by Act 123, 1972) provides for the reclamation of lands subjected to the mining of coal, gypsum, stone, and metallic ore (but not sand and gravel, clay, or peat). The effects of mining on water pollution is one of the objectives of a comprehensive survey mandated by the act. This survey is currently under way and is expected to result in promulgation of rules governing (among other things) pollution of public waters. Supervisor of the Act is the chief of the Geological Survey Division of the Department of Natural Resources. Completion of the comprehensive survey is hoped for in 1974.

The Water Resources Commission Act (Act 245, 1929, as amended, effective April 15, 1973) creates a Water Resources Commission "to protect and conserve the water resources of the state, to have control over the pollution of any waters of the state and the Great Lakes . . ." etc. Mining operators must obtain permits to discharge wastes into public waters.

Michigan's Mineral Well Act (Act 315, 1969) is intended "to provide control of the drilling, operating and abandoning of mineral wells to prevent surface and underground waste; . . . to provide for inspecting, repairing and plugging of mineral wells; . . ." etc. Supervisor of the Act is the chief of the Geological Survey Division. Full enforcement of this Act should go far toward controlling pollution from mineral wells.

Material in this and the following paragraph is taken from Deshpande and Dworsky (1973). Official Michigan policy states, "The business of mining and beneficiating low grade iron ore . . . is declared to be in the public interest and necessary to the public welfare, and the acquisition of private property for development of an adequate water supply, the necessary storage, processing and treatment of liquid and solid wastes . . . is declared to be for a public purpose." The Michigan Department of Natural Resources is authorized to acquire by condemnation land for settling ponds, storage basins and treatment, transportation and other facilities -- this land to be leased to the mine operator if he has already secured 75 percent of the necessary land (Michigan Statutes 425.171, P. A. 1968, no. 314).

"It is the policy of the State to develop and continue mining in the Upper Peninsula. It is in the public interest and for the public welfare." Because beneficiation needs water, the use of water in the processing of low grade iron ore is declared to be in the public interest (Michigan Statutes 323.251, sec. 1, P. A. 1959, no. 143).

3.9.2 Ohio

The surface waters of Ohio are protected under Regulation EP-1 of the Ohio Environmental Protection Agency, effective July 27, 1973, entitled "Water Quality Standards." Sources of water pollution are regulated by applying effluent standards and issuing permits, under the general style of the Federal Water Pollution Control Act, Section 402, "National Pollution Discharge Elimination System."

Regulations under Ohio's oil and gas law (Revised Code Chapter 1509) prohibit the contamination of surface or ground waters by disposal of oil-field brines, but drilling operations are so numerous and state regulatory agencies so undermanned that adequate control is not exercised.

Ohio has no other pollution-control program for its subsurface waters.

3.9.3 New York

New standards to control stream pollution became effective in New York on March 27, 1974. These are set forth in the State Department of Environmental Conservation's "Classifications and Standards Governing the Quality and Purity of Waters of New York State." Title 6, Official Compilation of Codes, Rules and Regulations, Part 702.1, refers to "Class A-Special (International Boundary Waters) (Great Lakes Water Quality Agreement of 1972)." Quality standards include, among other items, requirements that total dissolved solids shall not exceed 200 mg/l, and that suspended solids that will be deleterious for any best usage will not be allowed. Legislation to support these measures comes under Article 17 of the Environmental Conservation Law (formerly Article 12 of the Public Health Law) (W. E. Loveridge, NYSDEC, personal communication and enclosures).

It may be significant that a comprehensive survey by a regional planning agency, with full attention to major sources of pollution, especially of ground water, does not mention deleterious effects of any extractive industries. This report is listed in the references under Genesee-Finger Lakes Regional Planning Board.

3.9.4 Wisconsin

In Wisconsin, according to Wirth et al. (1973), "current state laws are adequate to control water pollution related to mineral resources." Deshpande and Dworsky (1973), on the other hand, find that the state lags behind Michigan and Minnesota in creating formal policies and safeguards against environmental damage that might be caused by mining. If mining operators are to divert waters for their use, they must secure a permit

from the Department of Natural Resources. "The Department shall impose such conditions in the permit as it finds are reasonably necessary in the public interest for the restoration of waters after completing of the mining operations or cancellation of the permit, for the orderly disposal of waste or tailings . . ." (Statute 107.05, sec. 9). Statute 144.01 subjects "unnecessary siltation" from quarries and gravel pits to regulation. Statute 29.29(3) prohibits discharge of deleterious substances into the state's waters. But all these regulations apply only if the mining operation requires diversion of water for its use. The regulations do not apply to ground water.

3.9.5 Minnesota

All state agencies in Minnesota are directed to cooperate with and assist the Pollution Control Agency, which was formed in 1967. The Division of Water, Soil and Minerals of the state's Department of Natural Resources is in charge of mining regulation. The commissioner of the Department may grant permits for drainage diversion and control or use of water where necessary for mining of iron ore, taconite, or copper -- if it is necessary and there is no other alternative, if it will not substantially impair the interests of the public, and if it will not endanger health or safety (Deshpande and Dworsky, 1973). If a permit is granted, the Commissioner will also prescribe the requirements for restoration of the waters to their former condition. This may require the mining operator to post a bond (Statute 105.64). Mine dewatering is monitored by the state's Conservation Department.

3.9.6 Other States

Only three Illinois counties drain into Lake Michigan. The state is currently limiting all discharges into the lake, and no new ones are to be allowed.

There are large sources of pollution in northern Indiana, but these are industrial plants rather than extractive industries. Urban runoff, industrial wastes, and waste-treatment processes are the significant sources of pollution of Lake Michigan and the streams tributary to it.

Only Crawford and Erie counties in Pennsylvania are tributary to Lake Erie. The state's "Clean Streams Act" and other legislation should be more than sufficient to control pollution from the single sand and gravel pit and the one peat bog reported to be active in this area.

Section 4

SUMMARY OF RESEARCH PROGRAMS

4.1 Nature of Studies

In the limited time available, it has not been possible to make a study of "current or proposed research, demonstration, or monitoring programs that may help answer basic questions." Such a study is desirable and should be made.

State-of-the-art studies of pollution produced by certain extractive industries are currently under way at the Robert S. Kerr Water Research Center of the U. S. Environmental Protection Agency, in Ada, Oklahoma. Reports are soon to be published on the significance of water pollution associated with the sand and gravel, uranium, and oil-shale mining industries. Research is also being done on sealants for mine-tailing sedimentation ponds. Studies bearing more directly on the extractive industries of concern in this review are in the discussion stage at the Center.

With a grant from the Federal Water Pollution Control Agency, researchers at the University of Minnesota and their associates have made a study of the role that peat might play in combating water pollution (U.S. Bur. Mines, 1972, p. 389).

The following news item appeared in the magazine Rock Products, July 1974.

> The Environmental Protection Agency finally is moving ahead with its previously announced study of mining effluents prior to establishing guidelines and new source performance standards. The consulting firm Versar, Inc., has been awarded the contract; it will meet with the trade associations to identify critical factors and review control methods currently in use. $3.5 million has been released to EPA by the Office of Management and Budget to participate in the joint U.S.-Canadian investigations set up by the Great Lakes Water Quality Agreement of 1972.

Section 5

NATURE AND AVAILABILITY OF TECHNOLOGY TO COPE WITH PROBLEMS

5.1 Nature of Technology

There is abundant published material on techniques of controlling pollution from mining activities (e.g., U.S. EPA Rept. EPA-430/9-73-011, 1973). Most of it deals either with acid-mine drainage and other problems of coal-mine areas, or with the treatment of waters laden with suspended solids from the milling of metallic ores. The former subject is of little significance in the Great Lakes region. The latter is applicable to the Minnesota iron-ore mines, to which indeed it has been intensively applied (Baillod et al., 1970, 1972). This technology, i.e. use of tailings ponds and related features, can be readily modified for use at gravel pits and stone quarries throughout the region if it is determined that the need exists.

When and if it is decided how best to dispose of unwanted brines, it will probably be found that existing techniques will do the job. Know-how is available to evaporate brines in order to recover their salts; to pump them into deeply buried strata via disposal wells (U. S. EPA Rept. EPA-430/9-73-012, p. 31-100); and to return them to oil-bearing strata to aid in repressuring and total ultimate recovery of oil. What is needed is fundamental research on alternative methods of disposal, especially on the long-range effects of deep-well injection.

Section 6

NEED FOR NEW PROGRAMS

6.1 Gaps in Knowledge

By far the largest extractive industries in the Great Lakes region are those that produce sand, gravel, and crushed stone. Large quantities of water are used in these operations, yet little information seems to exist on the extent of this use or on the degree to which contaminated waters are released. A systematic study, perhaps undertaken with the aid of the various states, would yield much-needed data on this subject. Such a study should be undertaken in the light of the expected expansion of these industries in the coming decades, as indicated on Table 3, page 15.

As previously indicated, there are appreciable gaps in our knowledge of the best way in which to dispose of waste brines. One aspect of the problem that should receive attention is a systematic inventory of those geologic formations beneath Michigan, Ohio, and New York that might act as large-scale receptors of brines. The rate at which these formations can accept introduced brines, and their ultimate storage capacity compared with the volume of brine expected to be produced over the next 40 to 50 years, are other topics on which information is needed.

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Category A5 - Pesticides and Herbicides

IMPACT OF AGRICULTURAL USE OF PESTICIDES ON THE WATER QUALITY OF THE GREAT LAKES

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To be used as portion of the U. S. Task A Report on GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by the International Joint Commission - prepared in partial fulfillment of U. S. Environmental Protection Agency Contract No. 68-01-1598

July 1974

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Section 1

SUMMARY

1.1 Introduction

The agricultural land in the United States portion of the Great Lakes Region is approximately 12,998,000 ha (32,118,000 acres), representing over 38% of the total land area of the region. More than 64% of the agricultural land is cultivated and devoted to: row crops (43%), hay and pasture (33%), small grains (17%), and specialty crops (7%). Farming activities are not well-distributed in the region. Crop production is concentrated in the agricultural watersheds of Lakes Michigan and Erie where more than 75% of the total cultivated cropland is located.

Pesticides are widely used in the region. Overall, more than half of the harvested area for crop production has been treated with pesticides. Herbicides and insecticides account for most of the pesticides applied, with herbicides being the predominant type of pesticide applied. Field crops receive the major treatment of pesticides. Among field crops, row crops--particularly corn--receive substantial treatment with pesticides, while hay and pasture receive the least. Orchards are treated extensively with insecticides. Some vegetable and processing crops, such as sweet corn and potatoes, also receive considerable insecticidal treatment.

All agriculturally-applied pesticides are potential pollutants to aquatic environments. The amount and nature of pesticides reaching the Great Lakes from agricultural watersheds are primarily functions of the persistence of the compounds used, intensity and length of time pesticides have been applied, and the applicable transport mechanisms. Quantification and identification of pesticide pollution originating from agricultural activities are difficult to assess because of the complexity of the watersheds. Limited information indicates that low-level concentrations of pesticides/metabolites, particularly the persistent types, may gain entry to tributary waters and the Great Lakes essentially through agricultural runoff.

1.2 Summary of Findings and Conclusions

Fragmentary inventories of the agricultural use of pesticides indicate extensive use of pesticides in the Great Lakes Region. Herbicides are the predominant pesticides used, and their use has been increasing in recent years. The use of organophosphorous and carbamate insecticides has increased because of the need to replace the majority of the persistent organochlorine insecticides.

Accumulations of pesticides--particularly the organochlorine insecticides--have occurred in agricultural soils as a result of repeated applications. Residues of pesticides or their metabolites may find their way to aquatic environments through the atmosphere, leaching to groundwater, and runoff. Agricultural runoff has been implicated as the primary avenue of pesticide transport to aquatic environments, and many investigations, mainly on small agricultural watersheds, have been undertaken on this aspect. Data from these investigations, in addition to limited data obtained from large watersheds, indicate that only low-level residues are transported to aquatic environments via runoff. Although the amounts of residues reaching the Great Lakes are in the ng/l (ppt, i.e., 1 part in 10^{12} parts) to µg/l (ppb, 1 part in 10^{9} parts) range, contamination can be magnified by the process of bioconcentration of residues in tissues of many aquatic organisms and by accumulation in bottom sediments.

Natural dissipation processes are operating to reduce or remove pesticide residues remaining in soils and aquatic environments. Microbial metabolism appears to be the major pathway of degradation for many pesticides in soils and sediments. Intensive studies have been made on this subject, but processes are still not clearly understood and degradation products of many pesticides are not fully identified. Additionally, the fate and toxicity of degradation products need continual assessment. Most herbicides are microbially degraded in soils and are much less persistent than the organochlorine insecticides. However, many of the commonly used compounds, such as atrazine, dichlobenil, diuron, monuron, picloram, trifluralin, simazine, paraquat and diquat, may persist for a year or longer in soils. Indications have been found that the persistence of many pesticides in aquatic environments differs from that in terrestrial soils, but

further investigations are needed to make generalizations.

The pesticide pollution of the Great Lakes, particularly Lake Michigan, is already well-documented. Pesticide input to the Great Lakes will continue as long as pesticides are used in the watersheds. Projections are that the cropland area in the U.S. portion of the Great Lakes Region will remain essentially unchanged in the next 50 years, but the human population will expand rapidly, thereby necessitating increased crop production per unit area in order to meet the food requirements of the population. To sustain high crop production, pesticides are essential. Because of this consideration, in addition to the absence of economically feasible alternative methods of pest control, pesticide usage in agriculture is predicted to increase. Use of less persistent pesticides will continue to rise due to the restrictions placed on the persistent organochlorine insecticides. Even with the use of less persistent pesticides, residue build-up may still result due to increasing use and repeated applications. Accumulation and impact of the degradation products of readily degraded pesticides in the environment are relatively unknown.

The Great Lakes States appear to have adequate existing laws and regulations concerning the sale, registration, distribution, use, and application of pesticides. The primary limitations for the success of pesticide regulations are inadequate funds and lack of needed personnel to do the monitoring and enforce the programs.

The existing voluntary basis for control of erosion and sediment build-up is extremely slow and, if allowed to continue at its current rate, abatement of pesticide pollution of the Great Lakes and related waterbodies will be limited in the next 50 years. Prevention of pesticide pollution of the Great Lakes can be greatly accelerated by the enactment and enforcement of mandatory erosion- and sediment-control programs by all levels of state governments concerned.

1.3 Recommendations

Research is needed urgently to adequately quantify and identify the contribution of agricultural activities to the pesticide pollution of the Great Lakes. Presently, only fragmentary information is available, and

the impact of agricultural usage of pesticides on the water quality of the Great Lakes can be only roughly estimated. Action should be taken on the following recommendations to provide meaningful information to assess the impact on water quality of pesticide usage for agricultural purposes.

a. Accounting of pesticide inputs to agricultural lands involving a systematic documentation of the kinds and amounts of pesticides used, category of crops treated, and area treated.

b. Evaluation of degradation rates of pesticides, particularly the newer compounds, including the formation, fate, and toxicity of their degradation products.

c. Investigations on the mechanisms of pesticide transport from varied soil types and cropping patterns present in the watersheds. Atmospheric inputs of pesticides to the Great Lakes should be investigated.

d. Expansion and redesigning of monitoring programs to include soil, sediment, and atmospheric samplings. Critical evaluation of sampling techniques, sample preservation, and analytical techniques should be made as often as possible.

e. Enactment of mandatory erosion- and sediment-control regulations by all levels of state governments. If regulations are already enacted, immediate enforcement is needed.

f. Better implementation of pesticide management techniques compatible with efficient soil and crop management systems must be practiced.

Section 2

SUPPORTING MATERIAL

2.1 Introduction

Agricultural activities in the Great Lakes Basin were initiated at the beginning of the 19th century, but the use of pesticides in crop production was virtually nonexistent prior to the mid-1940's. For the past three decades pesticide usage has vastly increased, making it possible to obtain the high yields of many crops never realized before. Pesticide use patterns indicate that the agricultural industry will not only continue to use pesticides, but it is predicted that they will be used at higher than present levels in order to sustain the higher crop production per unit area needed to meet the food demands of the rapidly expanding population around the Great Lakes. Since the early 1960's public concern has been expressed that pesticide residues--particularly the organochlorine insecticides--are widely distributed in the biosphere and have accumulated in several segments of the environment including the Great Lakes. Farming operations, which provide the major use for pesticides, are frequently cited for the pesticide contamination of the aquatic environment. These suspicions, however, have not been fully documented, and an attempt is made in this review to evaluate the present and future impact on the water quality of the Great Lakes from pesticides used on agricultural land.

2.2 Scope of Study

Data were obtained from published papers, reports, symposia proceedings, framework studies, abstracts, texts, and personal communications. References from the period of 1960 to 1974 were considered, but most of the literature surveyed was published and/or reported in the late 1960's and early 1970's. Although much of the information included was gathered outside the Great Lakes Basin, a fair amount of information is available for the basin and is given emphasis in the report.

2.3 Study Procedure

The literature evaluation on the sources, forms, and amounts of pesticide pollutants arising from agricultural activities was conducted under the direct supervision of the Director of the University of Wisconsin Water Resources Center (G. Chesters). A graduate student (G. V. Simsiman) assisted in compiling and summarizing needed data and wrote the preliminary report. Final review and revision of the draft submitted to the Great Lakes Basin Commission were done by the Director.

Information was gathered by writing many scientists, institutions, and government agencies, requesting copies of reprints, preprints, reports, and other kinds of information. The search for ongoing research on pesticides was undertaken utilizing the services of the University of Oklahoma-GIPSY computer program which is supported by the Office of Water Resources Research of the U.S. Department of the Interior. The major sources of literature/information used were:

1. Journal of Environmental Quality
2. Pesticide Monitoring Journal
3. Soil Science Society of America Proceedings
4. Journal of Economic Entomology
5. Journal of Agricultural and Food Chemistry
6. Weed Science
7. Weed Research
8. Environmental Science and Technology
9. Water Resources Bulletin
10. BioScience
11. Science
12. Water Pollution Control Federation Journal
13. Agronomy Journal
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28. Proceedings of Conferences on Great Lakes Research

29. Great Lakes Basin Framework Study

30. International Joint Commission Annual Reports

31. Selected Water Resources Abstracts

32. Water Resources Research Catalogs

33. USDA-CSRS Current Research Information Service

34. USDI, OWRR, Water Resources Scientific Information Center

35. USDA Agricultural Stabilization Service

36. USDA Economic Research Service

37. U.S.EPA Ecological Research Series

38. U.S.EPA Water Pollution Control Research Series

39. U.S.EPA Environmental Protection Technology Series

40. U.S.EPA Office of Pesticide Programs

41. Water Resources Center, Purdue University

42. Water Resources Center, University of Wisconsin-Madison

43. Water Resources Information Program, University of Wisconsin-Madison

44. Department of Entomology, University of Wisconsin-Madison
45. Michigan Department of Natural Resources
46. Wisconsin Department of Agriculture
47. Wisconsin Department of Natural Resources
48. Department of Soil Science, University of Wisconsin-Madison
49. Water Chemistry Degree Program, University of Wisconsin-Madison

2.3.1 Assumptions

Pesticide contamination of the Great Lakes has been revealed by state and federal monitoring programs. However, there are very few well-documented investigations on the sources, forms, and amounts of pesticide pollutants from various segments of the watersheds, particularly those from agricultural lands. The following assumptions are made:

a. The use of pesticides is inevitable and will continue to increase, particularly in the case of herbicides, because of the present lack of feasible alternatives for pest control.
b. Increased production per unit area on the essentially unchanged area devoted to crop production in the Great Lakes Region is necessary to meet the food requirements of the rapidly expanding population therein. To produce such high crop yields more and better production inputs, including pesticides, are needed.
c. Accumulated residues of pesticides, resulting from repeated applications to agricultural soils, serve as reservoirs for potentially transportable residues.
d. Soil erosion is a continuous geologic process but is accelerated by farming activities. Agricultural runoff, laden with pesticides, eventually reaches surface waters and provides low-level concentrations of residues. Abatement of soil erosion is proceeding only slowly and in a limited fashion.
e. The impact on water quality of agricultural use of pesticides is great in the Great Lakes Basin due to the extensive farming activities in the basin and the vast water surface area involved.
f. Agricultural use of pesticides in the Great Lakes Basin produces similar processes and relationships to those reported in other areas of the nation.

Section 3

REVIEW OF FINDINGS OF PREVIOUS STUDIES

3.1 General Description of Agricultural Land Use Category

3.1.1 Agricultural Activities

The Great Lakes Region consists of a total land area of 33,824,200 ha (83,579,700 acres) (66). In 1967 there were 11,579,000 ha (28,612,000 acres) of cropland; 1,419,000 ha (3,506,000 acres) of pasture land; 16,035,900 ha (39,624,700 acres) of forest land; 2,827,900 ha (6,987,700 acres) of urban and built-up areas; and 1,963,800 ha (4,852,500 acres) of other land. Cropland, including pasture, comprises the agricultural sector, and this sector represents more than 38% of the land area of the region. Forestry is practiced on 47% of the land; urban and built-up areas cover 8.4%; while 5.8% of the land is given over to other uses.

A wide variety of crops is grown in the region, including small grains, row crops, hay, vegetables, and fruits (Table 1). Estimates indicate that crop production is sufficient to supply the food and feed requirements of the population and the livestock and dairy industry of the region. The total cultivated cropland in 1967 comprised 8,361,700 ha (20,661,700 acres) or over 64% of the total agricultural land (includes cultivated cropland, pasture, and idle cropland). Over 43% of the cultivated cropland is devoted to row crops; 33% to hay and pasture; 17% to overall grains; and 7% to specialty crops (Table 2).

In order to sustain efficient crop production, good management practices are needed. Pest and disease control by pesticides has become indispensable in many areas of crop production.

Farm types and their location in certain geographic areas are related to climate, soils and markets. Farming activities appear to be concentrated in the Lake Michigan and Lake Erie regions (Table 2). About 75% of the total area devoted to producing all crop categories lies in these regions

TABLE 1

Land area and crop production in metric and English units for 1967 and predictions for 1980, 2000 and 2020 for the U.S. Great Lakes Region. (Recalculated from Appendix 13, Land Use and Management Work Group of the Great Lakes Basin Commission sponsored by the Forest Service and Soil Conservation Service of the U.S. Department of Agriculture (66).)

Crop	1967				1980				2000				2020			
	Area		Production		Area		Production		Area		Production		Area		Production	
	Hax10^3	Ax10^3	MTx10^3	Unit**x10^3	Hax10^3	Ax10^3	MTx10^3	Unit**x10^3	Hax10^3	Ax10^3	MTx10^3	Unit**x10^3	Hax10^3	Ax10^3	MTx10^3	Unit**x10^3
Wheat	710	1,756	1,773	65,142 Bu	797	1,970	2,624	96,400 Bu	754	1,862	3,044	111,800 Bu	796	1,967	3,637	133,600 Bu
Oats	686	1,696	1,454	100,135 Bu	756	1,868	2,241	154,375 Bu	649	1,604	2,083	143,500 Bu	423	1,046	1,536	105,813 Bu
Rye	24	60	44	1,739 Bu	24	59	48	1,902 Bu	28	69	64	2,516 Bu	35	87	86	3,386 Bu
Barley	18	45	45	2,089 Bu	39	96	129	5,917 Bu	35	86	132	6,042 Bu	29	71	122	5,625 Bu
Misc. Small Grains	17	43														
Corn for Grain	1,768	4,369	8,887	349,759 Bu	1,469	3,630	9,740	383,357 Bu	1,606	3,969	12,874	506,679 Bu	1,965	4,856	17,152	675,071 Bu
Corn for Silage	494	1,221	13,577	14,962 Ton	363	897	14,858	16,374 Ton	409	1,010	19,877	21,904 Ton	518	1,279	26,236	28,912 Ton
Soybeans	1,054	2,604	1,781	65,426 Bu	1,387	3,428	2,925	107,440 Bu	1,382	3,416	3,676	135,048 Bu	1,524	3,766	4,742	174,176 Bu
Dry Edible Beans	306	756	288	6,352 CWT	295	730	516	11,376 CWT	331	819	682	15,029 CWT	402	993	915	20,171 CWT
Sugar Beets	52	128	1,451	1,599 Ton	90	222	2,920	3,218 Ton	98	241	4,791	5,280 Ton	105	259	7,421	8,178 Ton
Potatoes	61	152	816	17,987 CWT	29	71	961	21,180 CWT	28	70	1,315	28,988 CWT	36	90	1,833	40,404 CWT
Fruits	243	600	1,002	1,104 Ton	161	398	1,323	1,458 Ton	165	407	1,903	2,097 Ton	185	456	2,719	2,996 Ton
Commercial Vegetables	211	520	2,091	46,093 CWT	213	526	3,284	72,380 CWT	226	558	4,505	99,295 CWT	267	659	6,224	137,171 CWT
Commercial Sod	21	53			21	53			21	53			21	53		
Alfalfa Hay	1,497	3,699	8,159	8,991 Ton	1,131	2,794	8,859	9,763 Ton	991	2,450	9,150	10,084 Ton	979	2,420	9,966	10,982 Ton
Clover-Tim-Other Hay	778	1,921	2,786	3,070 Ton	538	1,330	3,072	3,385 Ton	433	1,070	2,812	3,099 Ton	363	897	2,739	3,018 Ton
Cropland Pasture	422	1,042			303	749	1,312 *	1,446 *	270	667	1,500 *	1,653 *	289	714	2,077 *	2,289 *
Idle Cropland	3,216	7,947	0	0	3,618	8,939	0	0	3,316	8,194	0	0	2,396	5,920	0	0
TOTAL CROPLAND	11,578	28,612			11,234	27,758			10,742	26,542			10,333	25,533		
Improved Pasture	378	934			368	909	1,744 *	1,922 *	353	872	2,007 *	2,212 *	340	839	2,180 *	2,402 *
Improvable Pasture	908	2,243			888	2,194	3,844 *	4,236 *	859	2,123	2,476 *	2,729 *	833	2,058	2,734 *	3,013 *
Non-Improvable Pasture	133	329			131	325	167 *	184 *	128	317	188 *	207 *	128	316	210 *	231 *
TOTAL PASTURE	1,419	3,506			1,387	3,428			1,340	3,312			1,301	3,212		
TOTAL AGRICULTURAL LAND	12,997	32,118			12,621	31,186			12,082	29,854			11,634	28,745		

*Alfalfa hay equivalents (tons)
**Bushel weights for wheat, oats, rye, barley, miscellaneous small grains, corn for grain, and soybeans are 60, 32, 56, 48, 56, 56, and 60 lb, respectively.

Ha = Hectare
A = Acre
MT = Metric Ton

TABLE 2

Total area cultivated and area cultivated by crop categories for 1967 and projections for 1980, 2000, and 2020 in the U.S. Great Lakes Region. (Modified and recalculated from Appendix 13, Land Use and Management Work Group of the Great Lakes Basin Commission, sponsored by the Forest Service and Soil Conservation Service of the U.S. Department of Agriculture (66).)

Region	1967		1980		2000		2020	
	$Ha x 10^3$	$A x 10^3$	$Ha x 10^3$	$A x 10^3$	$Ha x 10^3$	$A x 10^3$	$Ha x 10^3$	$A x 10^3$
				TOTAL CROPLAND*				
Lake Superior	156.9	392.6	117.1	289.4	83.0	205.2	83.2	205.5
Lake Michigan	3,681.7	9,097.5	3,422.9	8,458.1	3,342.5	8,259.3	3,614.8	8,932.2
Lake Huron	825.2	2,039.1	810.1	2,001.8	875.1	2,162.3	1,007.8	2,490.2
Lake Erie	2,726.3	6,736.6	2,574.1	6,360.7	2,548.4	6,297.1	2,612.9	6,456.4
Lake Ontario	969.5	2,395.7	691.6	1,709.0	592.2	1,463.4	618.7	1,528.7
TOTAL	8,361.7	20,661.7	7,615.9	18,819.0	7,441.2	18,347.3	7,937.2	19,612.9
				SPECIALTY CROPS**				
Lake Superior	3.8	9.4	1.5	3.6	1.2	2.9	1.3	3.2
Lake Michigan	283.5	700.6	226.4	559.5	240.9	595.2	279.8	691.4
Lake Huron	46.5	115.0	70.2	173.5	79.0	195.2	90.6	223.8
Lake Erie	136.5	337.3	118.0	291.5	96.3	238.0	137.8	340.4
Lake Ontario	94.8	234.3	142.0	351.0	77.5	191.5	84.2	208.2
TOTAL	566.8	1,400.7	492.7	1,217.4	516.2	1,275.5	592.8	1,464.8
				ROW CROPS**				
Lake Superior	1.4	3.4	0.77	1.9	1.2	3.1	2.3	5.6
Lake Michigan	1,506.0	3,721.3	1,447.0	3,575.5	1,522.4	3,761.9	1,848.2	4,567.0
Lake Huron	398.1	983.7	361.6	893.4	413.7	1,022.2	518.5	1,281.2
Lake Erie	1,532.0	3,785.7	1,571.9	3,884.2	1,670.4	4,127.7	1,887.2	4,663.2
Lake Ontario	184.9	457.0	132.8	328.2	121.1	299.2	152.7	377.3
TOTAL	3,622.0	8,949.9	3,514.4	8,684.0	3,729.0	9,214.4	4,408.9	10,894.4
				SMALL GRAINS**				
Lake Superior	18.6	46.0	24.4	60.4	12.3	30.3	12.2	30.2
Lake Michigan	646.2	1,596.9	753.2	1,861.3	712.9	1,761.7	666.5	1,646.9
Lake Huron	153.6	379.5	202.8	501.0	197.6	488.3	190.1	469.8
Lake Erie	476.9	1,178.5	468.2	1,157.0	210.2	519.5	288.7	713.4
Lake Ontario	161.2	398.4	166.9	412.4	129.5	320.0	125.3	309.7
TOTAL	1,456.6	3,599.3	1,615.9	3,992.8	1,465.4	3,621.1	1,283.0	3,170.4
				HAY AND PASTURE**				
Lake Superior	135.1	333.8	90.4	223.5	68.4	168.9	67.4	166.5
Lake Michigan	1,245.7	3,078.2	996.3	2,461.8	866.2	2,140.5	820.3	2,027.0
Lake Huron	227.0	560.9	175.6	433.9	184.8	456.7	208.6	515.4
Lake Erie	580.8	1,435.1	416.0	1,028.0	331.2	818.5	300.3	742.0
Lake Ontario	528.4	1,305.8	315.3	779.1	264.1	652.7	256.4	633.5
TOTAL	2,716.2	6,711.8	1,993.0	4,924.8	1,714.4	4,236.3	1,652.5	4,083.3

*Excludes idle cropland

**Specialty crops: fruits, vegetables, potatoes, sugar beets; Row crops: corn, corn silage, soybeans, dry beans; Small grains: wheat, oats, barley, rye, misc. small grains; Hay and pasture: alfalfa, clover-timothy, cropland pasture, sod.

of Lakes Michigan and Erie. Based on the total cultivated cropland in 1967, crop production in terms of area in the five lake regions is in the order: Lake Michigan (43%) >Lake Erie (33%) >Lake Ontario (12%) >Lake Huron (10%) >Lake Superior (2%). Higher intensity of crop production in the Lake Michigan and Lake Erie regions implies that more pesticides have been used in these regions than in other lake regions.

3.1.2 Pesticide Use Patterns

In assessing the extent by which pesticides contaminate the environment it is important to determine how much has been used. Estimates of quantities of pesticides used by farms in 1966 (183) indicated that 160,000,000 kg (353,000,000 lb) of active ingredients were used for crops, livestock, and other purposes, representing 51% of the total quantity used in the United States which was 310,000,000 kg (681,000,000 lb). Over 149,000,000 kg (328,000,000 lb) were used for crops alone, which includes all crops, pasture and rangeland.

According to the Census of Agriculture, 1969 (184) about 16,000,000 ha (40,000,000 acres) of agricultural land in the eight Great Lakes States were treated with pesticides, comprising 28% of the total area treated in the United States. Pesticide usage in the Great Lakes Region and neighboring areas indicated that herbicides are the leading types of pesticides (>50%) used in treated areas followed by insecticides (183, 184, 205). Use of other pesticides, such as nematocides, fungicides and defoliants, are not as widely used as herbicides and insecticides. Herbicide use has been rapidly increasing, while use of other pesticides has not changed substantially.

An inventory of the kinds and quantities of pesticides used in the Great Lakes Region was first attempted in 1966 (43). According to this survey, 12,421,960 kg (27,378,000 lb) of pesticides (active ingredients, including petroleum and sulfur) were applied, with crops (all crops, pasture and rangeland) receiving the largest share (86%), followed by livestock (12%), and other purposes (2.0%). The 9,431,942 kg (20,788,000 lb)

of pesticides used in the region, exclusive of petroleum and sulfur, represent 6.3% of the 149,000,000 kg (328,000,000 lb) used on crops in the United States. A vast array of chemicals used on crops is listed in Tables 3, 4, 5 and 6. It is significant to note from the list that during this period herbicides constituted 51% of the total used, followed by insecticides (32%), fungicides (15%), and miscellaneous pesticides (1.7%). Of the total insecticides applied, the organochlorine insecticides (57%) were predominant over the organophosphorous compounds (20%) and carbamate insecticides (11%). The most widely used organochlorine compounds were aldrin, DDT, heptachlor and toxaphene. Corn was the major recipient of herbicides and insecticides. Apple and vegetable crops received substantial amounts of insecticides and fungicides.

Pesticide surveys were conducted after 1966 in five Great Lakes States on field crops (205) and in Wisconsin on fruits, vegetables, and processing crops (108). These surveys were not specific for the Great Lakes Region; however, they reflect changes in usage of pesticides throughout the area, particularly in Michigan and Wisconsin.

The study concerning the general farm use of pesticides was recommended by a task force created by the Governors of Illinois, Indiana, Michigan, Minnesota and Wisconsin after a conference in April, 1969 to review the pesticide situation at that time as it related to possible pesticide pollution. Primary emphasis was placed on the major field crops which account for the largest share of pesticide usage. The survey was conducted for 3 years (1969 to 1971) in all states of the group except Indiana which did not conduct the survey in 1971. In 1971, approximately 18,900,000 ha (46,800,000 acres) of field crops were treated with pesticides in the four lake states, an increase of 18% over 1969. Herbicides accounted for 79% of the treated area, with insecticides applied to nearly 20%. Less than 1% of the area received fungicides. Herbicide usage increased by 28% from 1969 to 1971, while use of insecticides decreased by 10%.

About 13,400,000 ha (33,200,000 acres) or 62% of the harvested area of 21,700,000 ha (53,600,000 acres) received herbicide and/or insecticide treatment at least once in 1971, which is 8% more than in 1969 (Table 7).

TABLE 3

Quantities used and area treated with selected fungicides, all crops, U.S. Great Lakes Basin, 1966.* (Adapted from Cywin and Ward (43).)

Fungicide**	Active Ingredients		Area Treated+	
	kgx10^3	lbx10^3	Hax10^3	Ax10^3
Inorganic				
Copper sulfates	11.4	25.2	4.4	11.0
Other coppers	57.3	126	14.4	35.6
Mercury compounds	2.6	5.8	11.7	28.8
Other inorganics	40.0	88.2	5.8	14.4
TOTAL INORGANICS (NOT INCLUDING SULFUR)	111	245		
Organic				
Dithiocarbamates				
Maneb	278	613	35.9	88.6
Zineb	382	842	87.2	215
Ferbam	70	154	18.5	45.8
Others	58	128	9.1	22.4
TOTAL DITHIOCARBAMATES	788	1,737		
Phthalimides				
Captan	395	871	59.9	148
Others	21	47	3.9	9.6
TOTAL PHTHALIMIDES	416	918		
Karathane, Dodine, Quinones	78.5	173	33.2	82.0
Phenols	7.0	15.4	2.4	6.0
Other organics	13.8	30.4	7.0	17.2
TOTAL ORGANICS	1,304	2,874		
TOTAL (NOT INCLUDING SULFUR)	1,415	3,119		
Sulfur	501	1,105	26.3	65.0
TOTAL FUNGICIDES	1,916	4,224		

*Estimates based on use shown by the ERS Pesticide and General Farm Survey, 1966.

**May include use for purposes other than as fungicides.

+Not additive since one or more ingredients or different commercial preparations of a single ingredient may be applied on same land areas.

TABLE 4

Quantities used and area treated with selected herbicides, all crops, U.S. Great Lakes Basin, 1966.* (Adapted from Cywin and Ward (43).)

Herbicide**	Active Ingredients		Area Treated†	
	kgx10^3	lbx10^3	Hax10^3	Ax10^3
Inorganic	110	243	19.6	48.4
Organic				
Arsenicals	1.5	3.4	2.4	6.0
Phenoxy				
2,4-D	1,202	2,650	1,681	4,153
2,4,5-T	21.1	46.6	45.0	111
MCPA	45.2	100	89.8	222
Other	2.0	4.4	9.6	23.8
TOTAL PHENOXY	1,270	2,801		
Phenyl urea				
Diuron	10.2	22.6	8.4	20.8
Linuron	116	256	78.0	193
Other	5.2	11.4	5.0	12.4
TOTAL PHENYL UREA	131	290		
Amides				
Propachlor	201	443	125	309
Propanil	---	---	---	---
NPA	67.6	149	48.0	119
TOTAL AMIDES	269	592		
Carbamates				
CIPC and IPC	35.0	77.2	32.5	80.2
CDAA	432	953	294	725
Other	126	278	37.8	93.4
TOTAL CARBAMATES	593	1,308		
Dinitro group	114	252	35.4	87.6
Triazines				
Atrazine	1,542	3,398	864	2,136
Propazine	---	---	---	---
Other	16.3	36.0	8.1	20.0
TOTAL TRIAZINES	1,558	3,434		
Benzoic				
2,3,6-TBA	249	549	118	292
Amiben	323	713	309	763
Dicamba	3.8	8.4	10.9	27.0
TOTAL BENZOIC	576	1,270		
Other organics				
Trifluralin	108	237	105	262
Others	103	226	63.0	156
TOTAL OTHER ORGANICS	211	463		
TOTAL ORGANICS (NOT INCLUDING PETROLEUM)	4,725	10,413		
TOTAL HERBICIDES (NOT INCLUDING PETROLEUM)	4,835	10,656		
Petroleum	573	1,264	24.8	61.4
TOTAL HERBICIDES (INCLUDING PETROLEUM)	5,408	11,920		

*Estimates based on use shown by the ERS Pesticide and General Farm Survey, 1966.

**May include use for purpose other than as herbicides.

†Not additive since one or more ingredients or different commercial preparations of a single ingredient may be applied on same land areas.

TABLE 5

Quantities used and areas treated with selected insecticides, all crops, U.S. Great Lakes Basin, 1966.* (Adapted from Cywin and Ward (43).)

Insecticides**	Active Ingredients		Area Treated+	
	kgx10^3	lbx10^3	Hax10^3	Ax10^3
Inorganics	358	790	22.1	54.6
Botanicals and biologicals	0.3	0.6	2.7	6.8
Synthetic organics				
Organochlorines				
Lindane	11.0	24.2	33.3	82.4
Strobane	---	---	---	---
TDE (DDD)	44.9	99.0	11.8	29.2
DDT	138	305	72.7	180
Methoxychlor	39.4	86.8	36.4	90.0
Endrin	0.5	1.0	1.0	2.4
Heptachlor	133	292	153	378
Dieldrin	16.2	35.8	20.6	51.0
Aldrin	1,245	2,744	1,033	2,552
Chlordane	26.0	57.2	18.9	46.6
Endosulfan	14.0	30.8	11.2	27.8
Toxaphene	47.0	104	26.1	64.4
Others	0.4	0.8	1.5	3.8
TOTAL ORGANOCHLORINE	1,715	3,780		
Organophosphorus				
Disulfoton	67.4	149	47.2	117
Bidrin	---	---	---	---
Methyl parathion	6.8	15.0	0.8	2.0
Parathion	110	242	128	317
Malathion	50.4	111	36.3	89.8
Diazinon	230	507	216	533
Trichlorfon	0.2	0.4	0.1	0.2
Azinphosmethyl	72.5	160	50.0	124
Ethion	32.7	72.0	14.1	34.8
Others	30.2	66.6	27.7	68.4
TOTAL ORGANOPHOSPHORUS	600	1,323		
Carbamates				
Carbaryl	306	674	82.3	203
Others	16.1	35.4	24.8	61.4
TOTAL CARBAMATES	322	709		
Other synthetic organics	1.1	2.4		
TOTAL SYNTHETIC ORGANICS	2,639	5,816		
TOTAL INSECTICIDES (NOT INCLUDING PETROLEUM)	2,997	6,606		
Petroleum	177	390	7.4	18.4
TOTAL INSECTICIDES	3,175	6,997		

*Estimates based on use shown by the ERS Pesticide and General Farm Survey, 1966.

**May include use for purposes other than as insecticides.

+Not additive since one or more ingredients or different commercial preparations of a single ingredient may be applied on same land area.

TABLE 6

Quantities used and areas treated with selected miscellaneous pesticides, all crops, U.S. Great Lakes Basin, 1966.* (Adapted from Cywin and Ward (43).)

Miscellaneous Pesticides**	Active Ingredients		Area Treated+	
	kgx10^3	lbx10^3	Hax10^3	Ax10^3
Miticides				
Dicofol	21.3	47.0	11.9	29.4
Chlorobenzilate	0.9	2.0	0.7	1.6
Aramite	---	---	---	---
Tetradifon	15.9	35.0	12.1	30.0
Others	6.2	13.6	7.7	19.0
TOTAL MITICIDES	44.3	97.6		
Fumigants				
Nemagon	10.0	22.0	0.1	0.2
D-D mixture	11.0	24.2	0.1	0.2
Sulfur dioxide	43.4	95.6	3.9	9.6
Others	49.9	110	27.0	66.8
TOTAL FUMIGANTS	114	252		
Defoliants and dessicants				
Arsenic acid	---	---	---	---
Magnesium chlorate	---	---	---	---
DEF and Folex	7.5	16.6	4.9	12.0
Others	---	---	---	---
TOTAL DEFOLIANTS AND DESSICANTS	7.5	16.6		
Rodenticides	0.8	1.8	3.0	7.4
Plant growth regulators				
Maleic hydrazide	14.5	32.0	1.8	4.4
Others	0.1	0.2	0.5	1.2
TOTAL PLANT GROWTH REGULATORS	14.6	32.2		
Repellents	2.8	6.2	1.1	2.6
TOTAL MISCELLANEOUS PESTICIDES	184	406		

*Estimates based on use shown by the ERS Pesticide and General Farm Survey, 1966.

**May include use for purposes other than those indicated.

+Not additive since one or more ingredients or different commerical preparations of a single ingredient may be applied on same land areas.

TABLE 7

Estimated areas of crops harvested and crops treated with herbicides for five Great Lakes States during 1969, 1970 and 1971.* (Recalculated from General Farm Use of Pesticides 1971 Wisconsin and Illinois, Michigan and Minnesota, Wisconsin Statistical Reporting Service, sponsored by the Wisconsin Department of Agriculture and U.S. Department of Agriculture (205).)

Harvested or Treated	Year	Illinois		%	Indiana		%	Michigan		%	Minnesota		%	Wisconsin		%	Total*		%
		Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated	Ha x10^3	A x10^3	Treated
								CORN											
Harvested	1969	4,039	9,980		1,983	4,901		673	1,662		1,999	4,939		1,079	2,666		7,789	19,247	
	1970	4,148	10,250		2,102	5,195		720	1,778		2,139	5,285		1,111	2,746		8,118	20,059	
	1971	4,229	10,450		--	--		902	2,230		2,644	6,533		1,233	3,048		9,009	22,261	
Treated	1969	3,413	8,433	84	1,727	4,267	87	585	1,445	87	1,524	3,765	76	832	2,055	77	6,353	15,698	82
	1970	3,510	8,673	85	1,807	4,464	86	642	1,587	89	1,706	4,216	80	967	2,390	87	6,826	16,866	84
	1971	3,722	9,196	88	--	--		824	2,036	91	2,231	5,513	84	1,064	2,630	86	7,841	19,375	87
								SMALL GRAINS+											
Harvested	1969	816	2,016		494	1,221		465	1,149		2,017	4,985		722	1,784		4,020	9,934	
	1970	666	1,645		442	1,093		444	1,097		1,969	4,866		719	1,777		3,798	9,385	
	1971	659	1,628		--	--		420	1,039		2,233	5,518		661	1,634		3,974	9,819	
Treated	1969	8	21	1	19	48	4	147	364	31	1,212	2,994	60	150	370	21	1,517	3,749	38
	1970	22	55	3	20	50	5	156	385	35	1,105	2,732	56	170	420	24	1,454	3,592	38
	1971	59	147	9	--	--		166	410	40	1,563	3,863	70	162	400	25	1,951	4,820	49
								SOYBEANS											
Harvested	1969	2,724	6,730		1,340	3,311		208	514		1,242	3,068		70	174		4,244	10,486	
	1970	2,752	6,800		1,340	3,311		212	524		1,266	3,129		62	153		4,292	10,606	
	1971	2,894	7,150		--	--		219	540		1,154	2,851		52	128		4,318	10,669	
Treated	1969	1,907	4,711	70	923	2,280	69	118	292	57	690	1,706	56	35	87	50	2,750	6,796	65
	1970	1,970	4,867	72	951	2,350	71	134	330	63	783	1,936	62	31	77	50	2,918	7,210	68
	1971	2,289	5,656	79	--	--		154	380	70	793	1,960	64	24	59	46	3,260	8,055	76
								HAY											
Harvested	1969	503	1,243		387	956		601	1,485		1,350	3,336		1,628	4,022		4,082	10,086	
	1970	510	1,260		377	932		577	1,425		1,308	3,231		1,625	4,016		4,019	9,932	
	1971	494	1,220		--	--		587	1,450		1,315	3,250		1,653	4,086		4,049	10,006	
Treated	1969	30	75	6	47	117	12	101	250	17	19	48	1.4	8	20	0.5	159	393	4
	1970	23	57	5	25	61	7	98	242	17	8	20	0.6	9	22	0.6	138	341	3
	1971	22	55	5	--	--		73	181	12	5	13	0.4	11	28	0.7	112	277	3
								MISCELLANEOUS CROPS++											
Harvested	1969	--	--		--	--		272	671		--	--		3	7		274	678	
	1970	--	--		--	--		239	590		165	407		4	9		407	1,006	
	1971	--	--		--	--		239	590		97	240		4	10		340	840	
Treated	1969	--	--		--	--		200	493	74	--	--		1	3	43	201	496	74
	1970	--	--		--	--		196	484	82	113	280	69	1	3	33	310	767	76
	1971	--	--		--	--		205	507	86	58	143	60	3	7	70	266	657	78
								TOTAL											
Harvested	1969	8,081	19,969		4,204	10,389		2,218	5,481		6,608	16,328		3,502	8,653		20,409	50,431	
	1970	8,076	19,955		4,262	10,531		2,191	5,414		6,847	16,918		3,521	8,701		20,635	50,988	
	1971	8,275	20,448		--	--		2,367	5,849		7,443	18,392		3,604	8,906		21,690	53,595	
Treated	1969	5,358	13,240	66	2,716	6,712	65	1,151	2,844	52	3,445	8,513	52	1,026	2,535	29	10,980	27,132	54
	1970	5,525	13,652	68	2,803	6,925	66	1,225	3,028	56	3,717	9,184	54	1,178	2,912	33	11,645	28,776	56
	1971	6,092	15,054	74	--	--		1,422	3,514	60	4,651	11,492	62	1,264	3,124	35	13,429	33,184	62

*Area treated with both herbicide and insecticide or more than once with either are counted only once.

**Does not include Indiana because of lack of data in 1971

+Includes oats, wheat, barley and rye

++Dry beans in Michigan, flax in Minnesota and tobacco in Wisconsin

Corn received 58% of the treatment, soybeans 24%, small grains 14%, and hay less than 1%. Large portions of the harvested area of corn (87%), soybeans (75%), and small grains (50%) were treated. Less than 3% of the hay area received any pesticide.

Atrazine, alone or in combination, and propachlor were the herbicides most commonly used on corn both for pre-emergence and post-emergence application (Table 8). The amine and ester formulations of 2,4-D were extensively used for post-emergence treatment of corn. Amiben was used on almost half of the soybeans treated. MCPA and 2,4-D amine were the leading herbicides applied to small grains.

Most of the insecticides were used on corn largely as pre-emergence treatments (Table 9). Only small portions of the soybean, small grain, and hay area were treated for insects. Aldrin, carbofuran, phorate, diazinon and buxten were the major insecticides used on corn. Major insecticides used for other field crops were of the organophosphorous and carbamate types.

A sharp decline in the usage of organochlorine insecticides was reported in the four Great Lakes States for 1971. Less than 1,400,000 ha (3,500,000 acres) of field crops (6% of the treated area) were treated, a decrease of 20% from 1970. Aldrin, used primarily on corn, topped all other organochlorine insecticides, with heptachlor and chlordane being used to lesser extents. The usage of DDT on field crops in 1971 continued to be minimal.

The use of insecticides on vegetables, fruits and crops for processing in Wisconsin was surveyed in 1969 (108). Approximately 83,925 ha (207,380 acres) or 51% of the total area of vegetables and crops for processing and 9,841 ha (24,317 acres) or 97% of the total fruit area were treated with insecticides. Of the 463,880 kg (1,022,385 lb) total insecticides applied in 1969, about 65% was used for vegetables and crops for processing. Sweet corn, potatoes, and apples received major portions of the insecticides.

In the Lake Michigan region of Wisconsin--which constitutes 31% of the insecticide-treated area of the state--approximately 177,092 kg (390,310 lb) or 38% of the total insecticides was used. A wide variety of chemicals were applied (Table 10). Lead arsenate, carbaryl, phorate, malathion, parathion, azinphosmethyl, and toxaphene, in that order, were the leading insecticides used.

TABLE 8

Rate of application and estimated area treated with major herbicides in four Great Lakes States for 1971. (Recalculated from General Farm Use of Pesticides 1971 Wisconsin and Illinois, Michigan and Minnesota, Wisconsin Statistical Reporting Service, sponsored by the Wisconsin Department of Agriculture and U.S. Department of Agriculture (205).)

Herbicide	Formulation*	Average rate of application per unit area**								Estimated area treated*							
		Illinois		Michigan		Minnesota		Wisconsin		Illinois		Michigan		Minnesota		Wisconsin	
		kg or liter	lb or qt	kg or liter	lb or qt	kg or liter	lb or qt	kg or liter	lb or qt	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3
CORN - PREEMERGENCE																	
All herbicides	WP	1.1	2.4	1.0	2.2	0.87	1.9	0.91	2.0								
	GP	3.3	7.3			4.0	8.7	2.1	4.7	3,310	8,178	507	1,252	1,339	3,308	735	1,815
	LC	1.5	1.6	1.2	1.3	1.4	1.5	1.6	1.7								
Atrazine	WP	1.0	2.2	1.1	2.3	0.82	1.8	0.96	2.1	788	1,946	353	872	349	862	523	1,293
Atrazine and propachlor	WP	1.9	4.2			1.1	2.5	1.4	3.0								
	GP	3.1	6.7			3.4	7.4			189	466			67	166	5.7	14
Atrazine and alachlor	WP	0.73	1.6	0.69	1.5	0.78	1.7	0.78	1.7								
	LC	1.8	1.9	1.4	1.5			1.6	1.7	126	311	92	227	30	73	105	260
Atrazine and sutan	GP	0.78	1.7	0.69	1.5	0.73	1.6	1.0	2.2								
	LC	1.9	2.0	1.7	1.8			2.4	2.5	361	891	8	20	16	39	3.6	9
Alachlor	WP	3.2	7.1			3.8	8.3	2.6	5.6								
	LC	1.5	1.6			1.6	1.7	1.6	1.7	397	981			83	206	60	148
Propachlor	WP	1.1	2.4					0.91	2.0								
	GP	3.4	7.5			4.0	8.7	3.0	6.6	1,056	2,609			783	1,934	4.0	10
2,4-D ester	LC	0.4	0.4	0.6						89	221	13	33				
EPTC and 2,4-D	GP	2.9	6.4							33	82						
CORN - POSTEMERGENCE																	
All herbicides	WP	1.0	2.2	0.96	2.1	0.87	1.9	0.87	1.9								
	GP					3.4	7.4										
	LC	0.4	0.4	0.5	0.5	0.5	0.5	1.0	1.1	1,382	3,415	395	975	1,523	3,764	431	1,065
Atrazine	WP	0.96	2.1	1.0	2.2	0.73	1.6	0.87	1.9	133	328	132	326	102	253	221	546
Atrazine and oil	WP	1.0	2.2	0.96	2.1	0.91	2.0	0.91	2.0	145	359	161	398	679	1,678	147	362
2,4-D amine	LC	0.4	0.4	0.5	0.5	0.5	0.5	0.6	0.6	442	1,093	45	112	446	1,103	36	88
2,4-D ester	LC	0.4	0.4	0.5	0.5	0.4	0.4	0.7	0.7	517	1,277	31	77	201	497	15	39
SOYBEANS																	
All herbicides	WP	0.82	1.8	0.78	1.7	0.82	1.8	0.82	1.8								
	GP	3.5	7.7	3.7	8.1	4.1	9.0	3.6	7.8								
	LC	1.2	1.3	1.6	1.7	1.1	1.2	1.5	1.6	2,286	5,648	153	378	792	1,598	23	58
Alachlor	GP	3.6	7.9	3.1	6.8	4.1	9.0	2.3	5.0								
	LC	1.4	1.5	1.7	1.8	1.3	1.4	2.3	2.4	336	830	7	18	115	283	4.5	11
Amiben	GP	3.5	7.7	3.7	8.2	4.1	9.0	3.8	8.4								
	LC	2.1	2.2	2.0	2.1	1.7	1.8	2.3	2.4	1,097	2,711	38	95	457	1,128	5.3	13
Linuron	WP	0.59	1.3	0.78	1.7	0.87	1.9	1.0	2.2	130	322	70	173	6	14	4.5	11
Trifluralin	LC	0.9	0.9	1.0	1.1	0.9	0.9	1.0	1.0	496	1,226	4	11	186	460	3.2	8
Alachlor and linuron	WP	0.50	1.1	0.64	1.4			0.50	1.1								
	LC	1.2	1.3	1.0				1.0	1.1	94	232	22	54			5.3	13
SMALL GRAINS																	
All herbicides	LC	0.5	0.5	0.5	0.5	0.4	0.4	0.5	0.5	38	93	119	295	1,562	3,859	159	393
2,4-D amine	LC	0.5	0.5	0.5	0.5	0.4	0.4	0.6	0.6	27	68	43	107	664	1,641	71	176
2,4-D ester	LC			0.5	0.5	0.4	0.4	0.6	0.6			28	70	89	221	32	78
MCPA	LC	0.5	0.5	0.5	0.5	0.4	0.4	0.5	0.5	6	15	43	106	726	1,794	54	135

*Before diluting, WP is weight of wettable powder; GP is weight of granular powder; LC is volume of liquid concentrate

** Where no rates are shown, no or insufficient reports were received to obtain conclusive data.

* Areas treated more than once are counted for each treatment.

TABLE 9

Rate of application and estimated area treated with major insecticides in four Great Lakes States for 1971. (Recalculated from General Farm Use of Insecticides 1971 Wisconsin and Illinois, Michigan and Minnesota, Wisconsin Statistical Reporting Service, Sponsored by the Wisconsin Department of Agriculture and U.S. Department of Agriculture (205)).

Insecticide	Formulation*	Average rate of application per unit area**								Estimated area treated*							
		Illinois		Michigan		Minnesota		Wisconsin		Illinois		Michigan		Minnesota		Wisconsin	
		kg or liter	lb or qt	kg or liter	lb or qt	kg or liter	lb or qt	kg or liter	lb or qt	Ha x10^3	A x10^3	Ha x10^3	A x10^3	Ha x10^3	A x10^3	Ha x10^3	A x10^3
CORN - PREEMERGENCE																	
All insecticides	WP	1.8	4.0					0.82	1.8								
	GP	2.9	6.3	2.2	4.9	3.3	7.3	3.1	6.9	2,221	5,488	110	271	673	1,663	338	835
	LC	1.1	1.2			0.57	0.6	0.76	0.8								
Aldrin	GP	2.5	5.5	2.0	4.3	2.3	5.1										
	LC	1.2	1.3			0.85	0.9			915	2,261	67	166	64	158		
Buxten	GP	3.4	7.4	3.5	7.7	3.8	8.4	3.2	7.0	224	554	5.7	14	138	341	32	79
Diazinon	GP	2.8	6.1	3.4	7.4	3.2	7.1	2.0	4.4	93	230	15	38	38	94	27	67
Heptachlor	GP	2.6	5.8			1.1	2.5			173	428	13	31	12	30		
Phorate	GP	3.0	6.6			3.0	6.6	2.8	6.2	376	928			72	179	81	201
Carbofuran	GP	3.7	8.1			3.5	7.8	3.5	7.7	289	713			240	593	172	424
CORN - POSTEMERGENCE																	
All insecticides	WP					1.6	3.5	0.73	1.6								
	GP	2.8	6.1			3.2	7.0	2.7	5.9	90	223	1.2	3	55	136	30	75
	LC					0.85	0.9										
Diazinon	GP					3.2	7.0	3.0	6.7					38	94	12	30
Phorate	GP	2.9	6.5					2.3	5.0	36	90					8.9	22
SOYBEANS																	
All insecticides	LC					1.4	1.5			52	129	1.6	4	4.9	12		
SMALL GRAINS																	
All insecticides	WP			0.64	1.4												
	LC			1.0	1.1			0.76	0.8	20	49	85	209	1.6	4	2.8	7
Carbaryl	WP			0.64	1.4							70	172				
Malathion	LC			1.0	1.1			0.76	0.8			8.5	21			2.4	6
HAY																	
All insecticides	WP			0.78	1.7			0.45	1.0								
	LC	1.5	1.6	1.3	1.4			1.2	1.3	21	51	68	168	1.6	4	3.2	8
Azinphosmethyl	LC			0.85	0.9							16	39				
Carbaryl	WP			0.78	1.7							19	48			1.2	3
Diazinon and methoxychlor	LC	2.0	2.1	1.5	1.6					12	30	10	25				
Malathion	LC			1.3	1.4							6.9	17				
Malathion and methoxychlor	LC			1.9	2.0							12	30			1.2	3

*Before diluting, WP is weight of wettable powder; GP is weight of granular powder; LC is volume of liquid concentrate

** Where no rates are shown, no or insufficient reports were received to obtain conclusive data.

*Areas treated more than once are counted for each treatment.

TABLE 10

Insecticides used in processing crops, fruit and vegetable production for the Lake Michigan region of Wisconsin in 1962 and 1969. Excerpted and recalculated from Libby, Wisconsin Commercial Fruit and Vegetable Production Crop Acreage and Insecticide Use Survey, Mimeo Rept. (107), Dept. of Entomology, Univ. of Wisconsin, College of Agriculture, and Libby and Koval, Wisconsin Commercial Fruit, Vegetable, and Processing Crop Acreage and Insecticide Use Survey, Resource Rept. 13 (108), University of Wisconsin-Extension.

Insecticide	1962				1969			
	Area treated*		Insecticide used		Area treated		Insecticide used	
	Ha	A	kg	lb	Ha	A	kg	lb
Aldrin	4,484	11,079	8,602	18,960	0	0	0	0
Azinphosmethyl	138	341	870	1,918	3,018	7,458	10,880	23,980
Bacillus thuringiensis	--	--	--	--	369	911	--	--
Carbaryl	1,015	2,508	16,175	35,651	4,871	12,037	38,503	84,864
Chlordane	20	50	38	83	223	550	628	1,385
Chloropropylate	--	--	--	--	53	131	150	331
DDT	8,266	20,426	110,143	242,768	1,727	4,267	9,095	20,046
Diazinon	190	470	843	1,857	1,028	2,541	9,343	20,592
Dieldrin	1,979	4,890	3,549	8,043	22	54	31	68
Demeton	18	45	10	23	240	594	79	175
Dimethoate	--	--	-	--	84	207	47	104
Disulfoton	--	--	--	--	1,710	4,226	4,566	10,063
Endosulfan	559	1,381	865	1,907	2,328	5,753	4,445	9,797
Endrin	1,744	4,310	916	2,019	0	0	0	0
Ethion	380	939	796	1,754	357	881	511	1,126
Gardona	--	--	--	--	225	557	2,594	5,718
Genite	69	170	309	680	0	0	0	0
Heptachlor	3.2	8	1.8	4	0	0	0	0
Kelthane	127	315	129	284	74	182	142	312
Lead arsenate	1,256	3,104	48,082	105,978	1,828	4,516	39,407	86,858
Malathion	2,452	6,059	26,239	57,834	2,077	5,133	13,738	30,280
Methoxychlor	798	1,971	20,774	45,788	96	236	1,102	2,429
Meta-SystoxR	--	--	--	--	47	115	26	58
Mevinphos	956	2,363	686	1,512	385	952	282	622
Methomyl	--	--	--	--	1,197	2,957	3,039	6,698
Mitox	49	122	146	322	0	0	0	0
Morestan	--	--	--	--	34	83	44	96
Naled	269	665	302	665	156	385	250	550
Ovex	191	473	344	758	0	0	0	0
Parathion	4,340	10,725	17,169	37,843	5,065	12,515	11,576	25,514
Phorate	2,013	4,975	5,876	12,952	5,688	14,055	16,283	35,890
Phosphamidon	40	100	224	494	243	600	1,250	2,755
Perthane	0	0	0	0	25	61	86	189
Rotenone	0	0	0	0	1.2	3	2.3	5
TDE (DDD)	475	1,174	6,211	13,690	89	221	714	1,573
TEPP	42	105	44	98	26	64	188	415
Tetradifon	218	538	461	1,105	74	183	126	277
Toxaphene	2,955	7,302	34,732	76,553	1,040	2,570	7,609	16,770
Trithion	43	106	37	81	5.3	13	7.3	16
VC-13	451	1,114	360	794	257	636	342	754
TOTAL	35,544	87,829	305,036	672,331	34,661	85,647	177,086	390,310

*Door County not included

A comparison between the survey in 1962 (107) and that in 1969 (108) revealed a decreasing trend in the area treated and total quantity of insecticide applied (Table 10). However, there was a significant shift in the kinds of chemicals used. The use of organochlorine insecticides, especially DDT, aldrin, methoxychlor, TDE (DDD), toxaphene, dieldrin and endrin,was either decreased drastically or discontinued entirely. This was accompanied by a rapid increase in the use of carbamate (particularly carbaryl) and organophosphorous (diazinon, phorate, disulfoton, and phosphamidon) insecticides.

It has been shown in the preceding discussion that pesticides are used widely in the Great Lakes Region. In one way or another, more than half of the harvested area for crop production has been treated with pesticides. Herbicides are the predominant pesticides used and their use has been increasing in recent years. Shifts to use of different insecticides are evident. The use of organophosphorous and carbamate insecticides has increased, while the use of organochlorine insecticides has declined sharply. Field crops receive most of the pesticides, particularly the herbicides and insecticides. Major proportions of the herbicides and insecticides have been used to protect row crops, i.e., corn and soybeans. Sweet corn, potatoes and apples received considerable insecticidal treatment.

The surveys mentioned, though not accurate, represent the best available data. Obviously, more comprehensive inventories of the kinds and amounts of pesticides used in the region are needed, especially since some organochlorine insecticides were placed on the restricted use list.

3.2 Length of Time of Pesticide Usage Practice on Agricultural Land

The use of pesticides dates back a hundred years, but usage did not increase tremendously until about 1945 when DDT was introduced commercially. A rapid growth of the organic pesticide industry followed for the next two and a half decades (184). Chemical control has revolutionized agricultural production, with the attainment of high crop yields never before realized. Almost all agricultural practices formerly used to combat weeds, insects, and diseases were shifted to favor the use of pesticides. Since the intro-

duction of DDT, a myriad of chemicals have been synthesized. Approximately 34,500 pesticide products are currently registered with EPA and USDA (67). Farmers use the largest volume of pesticides, but about one half of the registered products are utilized by persons other than farmers.

DDT was believed to be used first in the Great Lakes watershed in the late 1940's (148). Although no comprehensive record exists of the types and amounts of pesticides used for various crops grown in the watershed, fragmentary evidence presented under Section 3.1.2 shows that pesticides have been used extensively, probably for the past 25 years. Types and amounts of pesticides used depend upon the crop grown. Field crops and orchards have been treated quite heavily while pasture land received little. Only in recent years have attempts been made to restrict the use of the persistent organochlorine insecticides and replace them with the less persistent organophosphorous and carbamate types of insecticides.

3.3 Types and Nature of Pesticide Pollutants

All agriculturally-applied pesticides are potential pollutants of aquatic ecosystems. The type and nature of pesticide residues reaching the Great Lakes from agricultural watersheds is controlled largely by: 1. amounts, intensities, and lengths of time pesticides have been applied; 2. persistence and/or residence time in the watershed; and 3. mobility of residues.

3.3.1 Insecticides

The organochlorine insecticides are the pesticides most studied from the standpoint of environmental contamination. Because of their long persistence in soils they provide a greater potential than other pesticides for contaminating aquatic systems. Furthermore, their low water solubilities make them strongly adsorbed to soil colloids, facilitating transportation by erosion. Numerous persistency studies indicate that the residence time of such commonly-used compounds as DDT, dieldrin, aldrin, chlordane, heptachlor, and lindane in soils is several years (45), and DDT is detected

frequently over a decade after application.

Use of organophosphorous and carbamate insecticides increased rapidly following restriction on the use of organochlorine insecticides in many Great Lakes States after 1969. Although organophosphorous compounds are more toxic to mammals than the organochlorine compounds, they persist for a relatively shorter length of time in soils (94). Similarly, the carbamates do not persist long in soils (158).

3.3.2 Herbicides

Herbicides are generally less ubiquitous in the environment than organochlorine insecticides. However, compounds such as s-triazine herbicides, picloram, monuron and related substituted ureas, and 2,4,5-T often remain in soils for at least a year after application (5). Atrazine is a widely-used herbicide for field crops, particularly corn. 2,4-D and the carbamate herbicides are short-lived in soils.

3.3.3 Fungicides

Unlike herbicides and insecticides, the fate of fungicides in soils is practically unknown. Major usage of fungicides occurs in orchards and vegetable farms, which abound in the Great Lakes watersheds. Mercurial fungicides have been a recent concern following the detection of mercury contamination of the Great Lakes in early 1970 (182). Since the use of mercury fungicides has been severely restricted, agricultural usage of mercury is negligible in comparison to industrial usage (154). In addition, the proportion of croplands treated with fungicides is small compared to the area treated with herbicides and insecticides. Considering these facts, it is doubtful that agricultural fungicides are contributing pollution to the Great Lakes.

3.3.4 Degradation Products

It is well known that the metabolites detected usually in the partial

degradation of DDT are DDE in aerobic systems and TDE (DDD) in anaerobic systems. DDE appears to be resistant to aerobic and anaerobic metabolism, thus its accumulation is possible in aquatic environments. Other pesticides may be metabolized to compounds equally or more active than the parent compound, as shown in the epoxidation of aldrin and dieldrin. Metabolites of many pesticides have not, as yet, been clearly identified, nor have their possible toxicity, persistence and accumulation in the aquatic ecosystem been defined adequately.

3.4 State-of-the-Art in Assessing and Quantifying Problems

The discussion of this section is covered extensively and is integrated in several sections, namely, 3.1.2, 3.5, and 3.6, and no further discussion is presented in this section.

3.5 Mobility of Pesticides in Soil, Groundwater, and Surface-Water Systems

Pesticides used to protect crops and pastures from weeds, insects, and diseases may enter the Great Lakes via: 1. the atmosphere, 2. groundwater, and 3. surface runoff. The discussion on pesticide entry will not be limited to information gathered in and around the Great Lakes Basin, but will include pertinent information obtained elsewhere.

A general scheme of the distribution and fate of herbicides in the biosphere is shown in Fig. 1 (55). Although the diagram was prepared for herbicides, it is equally applicable to other pesticides and is entirely self-explanatory.

3.5.1 Pesticide Movement to the Atmosphere

The atmosphere becomes contaminated with pesticides by: 1. drift during application, 2. volatilization, direct or from soil and other surfaces, and 3. wind erosion. That the atmosphere is contaminated with pesticides is well-documented (52).

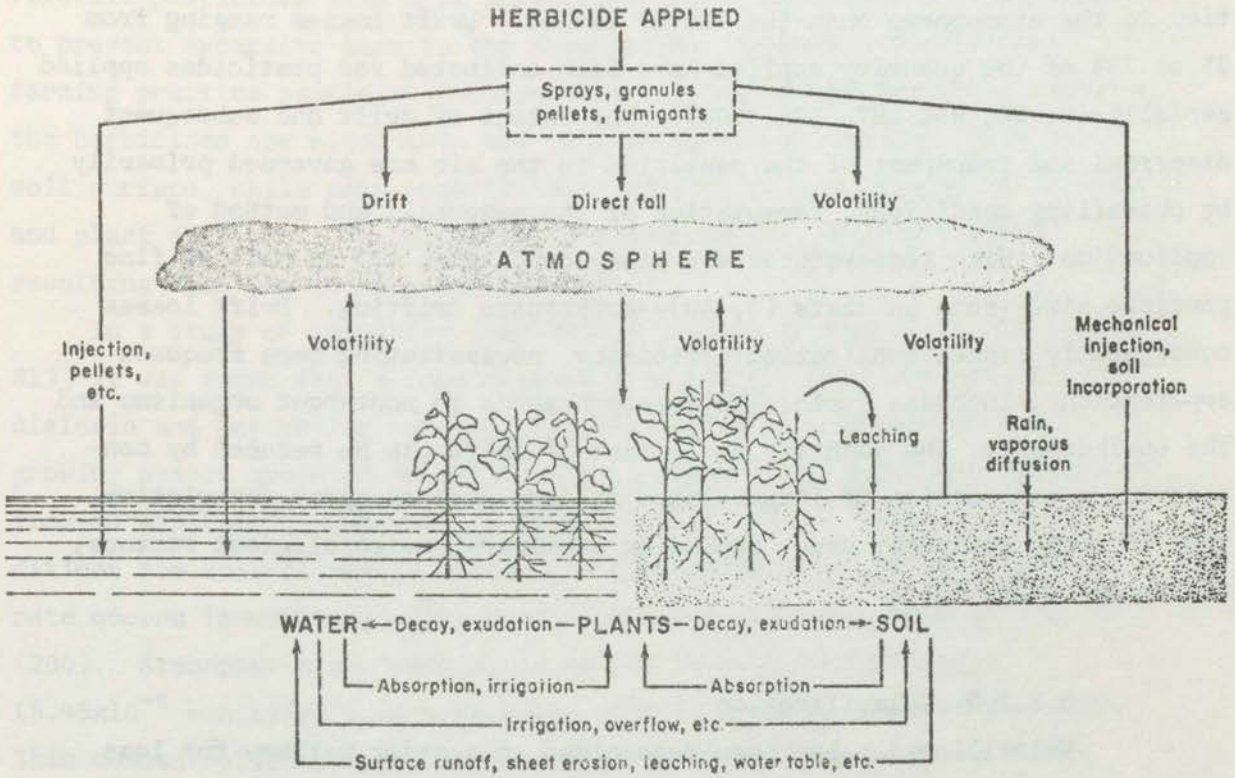

Fig. 1 Scheme showing the possible distribution and fate of herbicides and their degradation products in the biosphere. Redrawn from Foy and Bingham (55).

3.5.1.1 Drift

Drift is that portion of spray that is moved away from the target area by wind or other meteorological factors. The importance of inadvertent losses from drifting sprays and dusts has been fully recognized (130). Aerial spraying, which is a more widely used method of application as opposed to ground application (124, 184), contributed more pesticide pollution to the atmosphere than the latter method. Drift losses ranging from 25 to 75% of the quantity applied have been estimated for pesticides applied aerially (4, 28, 85, 187, 188, 206). The extent of drift and subsequent dispersal and transport of the pesticide in the air are governed primarily by prevailing conditions, formulation of the material, and method of application. High temperatures and windy conditions (3) as well as fine particle size--such as dusts (3, 60)--accentuate drifting. Drift losses consequently reduce application efficiency, necessitating more frequent applications, increase costs, and create hazards to nontarget organisms and the environment. The quantity and extent of drift can be reduced by considering a combination of interrelated factors such as spray formulation, type of spray equipment used, including atomizers, meteorological factors, and spray method (118).

3.5.1.2 Volatilization

Volatilization has been recognized as a major pathway for loss of pesticides from soil, plant and water surfaces. The factors and mechanisms of volatilization losses in soils are discussed in a comprehensive review (177).

After volatilization related to application has occurred, the remaining pesticides ultimately reach the soil. In soil, loss is evident for both surface-applied and soil-incorporated pesticides. The vaporization rate of a pesticide is related to its vapor pressure, but once it is in contact with the soil its vapor pressure is modified by environmental variables. Climatic and edaphic factors regulating the volatilization rate include: air movement (29, 51, 78), temperature (51, 70, 78, 173, 202), relative humidity of the surrounding air (11, 87), soil moisture content (78, 87, 97,

142, 173, 175, 176, 200, 202), soil organic matter content (2, 70, 143, 173) and pesticide concentration in the soil (51, 175).

Field measurements indicate that significant volatilization loss may occur if pesticides are not incorporated in the soil. Losses have been observed for DDT (85, 200), TDE (DDD) (200), dieldrin (30, 176, 202), endrin (201), TDE (200), lindane (174, 175, 176), heptachlor (31), and IPC and CIPC (142). It appears that incorporation of some of the more volatile pesticides into soil immediately after application is necessary to prevent excessive loss to the atmosphere. However, under normal farming practice pesticide incorporation is not always possible. Many of the herbicides are mixed into the soil or banded in the row below the soil surface, while many insecticides and fungicides are applied on soil and plant surfaces. The potential for atmospheric pollution exists, resulting from pesticide volatilization.

In a study of pesticide loss from a watershed cropped to corn (30, 31), it was found that a considerable fraction of soil-incorporated dieldrin and heptachlor was lost by volatilization. Losses in one growing season amounted to 2.8-2.9% for dieldrin and 3.9% for heptachlor from an application rate of 5.6 kg/ha (5.0 lb/acre). Under field conditions the volatilization process is continuous although the highest rate occurs immediately after pesticide application as shown in Fig. 2 (200). Atmospheric concentrations of DDT as high as 2,040 ng/m^3 (5.45×10^{-8} $ounces/yd^3$) were measured initially above a nonflooded plot. This concentration dropped to 100 ng/m^3 (2.67×10^{-9} $ounces/yd^3$) 2 days after application. Corresponding levels of TDE (DDD) decreased from 575 to 92 ng/m^3 (1.54×10^{-8} to 2.45×10^{-9} $ounces/yd^3$) within the first 2 days. Except for the first few days after application, pesticide concentrations seldom exceeded 100 ng/m^3 (2.67×10^{-9} $ounces/yd^3$), and subsequent change in atmospheric concentrations during a 6-month period appeared to be related to climatological factors.

3.5.1.3 Wind Erosion

Pesticides dispersed in the atmosphere become associated with airborne particulate matter and, as such, are capable of being transported

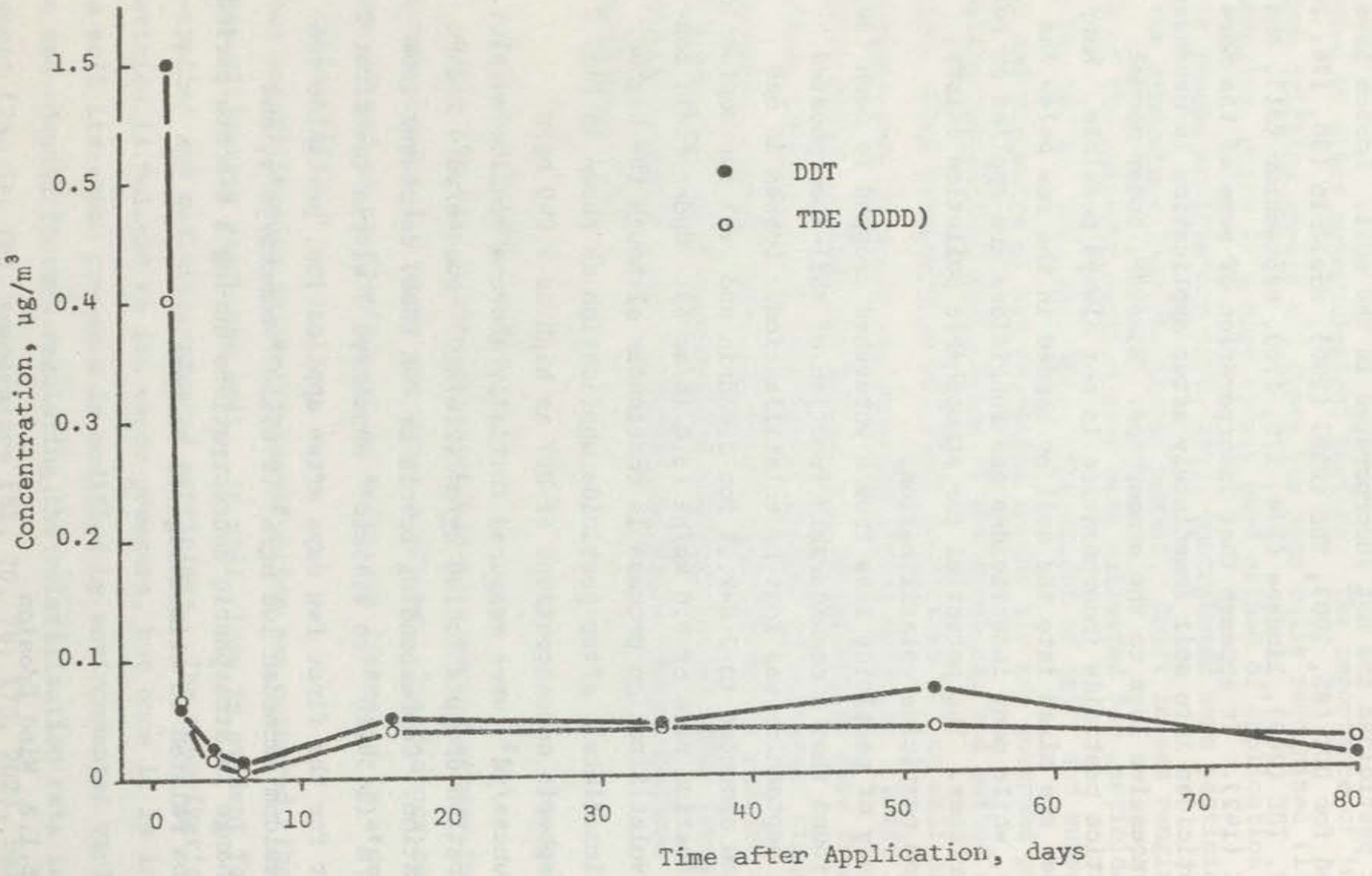

Fig. 2. Atmospheric concentration of DDT and TDE (DDD) monitored at 30 cm above the soil surface. Figure assembled from data of Willis et al. (200). $\mu g/m^3 = 2.67 \times 10^{-2}$ ounces/yd^3

(38, 194). Their distribution away from application sites is dependent on the prevailing wind movement and rate of fallout. Pesticide-laden dust may also originate from dislodgement of the soil-pesticide complex due to wind erosion (38, 121). Wind erosion may provide an important atmospheric source of pesticides when the pesticides are redeposited on watersheds and waterbodies.

Earlier discussions show ample evidence of atmospheric contamination by pesticides from drifting spray, by air-borne particulate matter, and through volatilization from soil surfaces. By these means, the potential for polluting areas not treated directly with pesticides is great when the pesticides are removed from the atmosphere through gravitational fallout and rain washout. Precipitated dust collected in Cincinnati after a dust storm indicated the presence of DDT, chlordane, Ronnel, and DDE as the major pesticide components of the dust (38). Heptachlor epoxide, 2,4,5-T and dieldrin were present in lesser amounts. The concentration ranged from 3 ng/g (3 ppb) dieldrin to 600 ng/g (600 ppb) DDT based on the air-dried weight of the dust. It was believed that the dust originated in the southeast of the United States where agricultural fields were treated heavily with pesticides. A country-wide attempt to determine atmospheric contamination by pesticides was performed by collecting samples in nine locations which included urban and rural sampling sites (178). Nineteen pesticides and metabolites were sought in the samples. Detected pesticide levels ranged from the lower level of detection of 0.1 ng/m^3 air (2.67×10^{-12} ounces/yd^3) to as high as 1,560 ng/m^3 (4.16×10^{-8} ounces/yd^3) *p,p'* DDT, 2,520 ng/m^3 (6.73×10^{-8} ounces/yd^3) toxaphene, and 465 ng/m^3 (1.24×10^{-8} ounces/yd^3) parathion. Only DDT was found in all localities. Levels of DDT in the atmosphere were higher in agricultural areas, and atmospheric content was more closely correlated with spraying activities than with rainfall.

Data revealed that DDT distribution in coastal and oceanic waters results from fallout of airborne particulate material (151). This implies that a vast area of water surface, such as the Great Lakes, may receive significant inputs of pesticides from the atmosphere (89). Normal levels of airborne pesticides could be low, but the return of these compounds

either in solution or in the adsorbed state to the earth's surface is continuous; however, their contribution to aquatic systems is difficult to evaluate (144). These compounds, if dissolved by rainfall, may gain entry to the aquatic system or be deposited on watersheds and become adsorbed by soil colloids and suspended materials.

The presence of pesticides in the atmosphere is well-documented. However, little is known about the ultimate fate of these compounds after they enter the atmosphere. More research is needed to evaluate the photochemical alteration, adsorption-desorption on dust particles, fallout and rainout of airborne pesticides.

3.5.2 Pesticide Movement to Groundwater

Pesticide contamination of groundwater can occur through leaching. Downward movement of agriculturally-applied pesticides is controlled by soil, pesticide, and climatic factors. These controlling factors are well-documented in reviews (1, 10, 82, 179, 190). Leachability of a compound depends primarily on the degree to which it is adsorbed to soil colloids. Adsorption is associated closely with organic matter content and occasionally with clay for nearly all pesticides. Pesticides are leached more readily in coarse-textured than in fine-textured soils as the latter contain more clay and generally more organic matter. Furthermore, water infiltration is relatively faster in coarser-textured soils. The solubility of pesticides plays an important role in their movement in the soil since solubility limits the concentration of the compound in the soil-water phase. Solubility--which is largely dependent on the chemical constitution of the compound--is shown to be correlated inversely with adsorptivity. Thus, pesticides of high water solubility are more subject to vertical movement than those of low water solubility. The transport of pesticides through the soil is conditioned by the amount, intensity and frequency of percolating water. Water facilitates desorption as well as dissolution of particulate or adsorbed compounds.

Adsorption of nonionic pesticides--which include the organochlorine and organophosphorous insecticides--is correlated primarily with soil organic matter content (2, 10, 16, 64, 68, 76, 80, 81, 86, 101, 143, 150)

and to a lesser extent with clay content. Retention of acidic and basic compounds is affected markedly by soil pH (44, 75, 79, 189, 192). Soil reaction controls the overall charge of the molecule and hence its adsorptivity to clay and organic colloids. The organic cations, diquat and paraquat, are held strongly by clay minerals and are often adsorbed irreversibly (193). Weakly-adsorbed water-soluble compounds are desorbed readily by water and hence pose a greater potential for leaching, depending upon the composition and amounts of the soil colloidal fractions.

Numerous studies have been conducted to determine the relative mobilities of pesticides in soils. The findings of these investigations are summarized in Table 11 (82). The organochlorine insecticides--which have limited water solubility--are the least mobile, followed by the organophosphorous insecticides. The water-soluble acidic herbicides are most mobile. Most of the pesticides, such as triazines, phenylureas and carbamates, have intermediate mobility. Within a diverse group of pesticides, relative mobility is related essentially to solubility.

Organochlorine insecticides are, in general, nonleachable. Field trials have shown that they are retained largely in the upper 15 to 20 cm layer of most agricultural soils (14, 33, 37, 46, 102, 112, 117, 133, 167, 180, 199, 203) including aquifers (159). Any movement to lower depths and subsequently to water tables is attributed by some investigators to be the result of the physical transport of adsorbed or particulate compounds by water through vertical cracks formed during dry periods (180, 199). Thus, holes made by soil animals may be important in the downward movement of pesticides.

Limited data on pesticide concentration in groundwater collected on treated watersheds and in wells indicate that low or negligible quantities of pesticides are transported through the soil profile (88, 90). However, in areas with a shallow and fluctuating water table, frequently found in coastal sandy soils (102) and aquifers (159), appreciable amounts of pesticide may find their way to the underlying water; this is especially true for the soluble herbicides. Toxaphene and fluometuron which reached groundwater were found by LaFleur et al. (102) to persist for a year.

In summary, the entry of a particular pesticide to groundwater is defined by soil type, depth to water table, rainfall infiltration and

TABLE 11

Relative mobility of pesticides in soils*
(Adapted from Helling et al. (82).)

Mobility Class**				
5	4	3	2	1
TCA+	Picloram	Propachlor	Siduron	Neburon
Dalapon	Fenac	Fenuron	Bensulide	Chloroxuron
2,3,6-TBA	Pyrichlor	Prometone	Prometryne	DCPA
Tricamba	MCPA	Naptalam	Terbutryn	*Lindane*
Dicamba	Amitrole	2,4,5-T	Propanil	*Phorate*
Chloramben	2,4-D	Terbacil	Diuron	*Parathion*
	Dinoseb	Propham	Linuron	*Disulfoton*
	Bromacil	Fluometuron	Pyrazon	Diquat
		Norea	Molinate	*Chlorphenamidine*
		Diphenamid	EPTC	Dichlormate
		Thionazin	Chlorthiamid	*Ethion*
		Endothall	Dichlobenil	*Zineb*
		Monuron	Vernolate	Nitralin
		Atratone	Pebulate	C-6989
		WL 19805	Chlorpropham	*ACNQ*
		Atrazine	*Azinphosmethyl*	*Morestan*
		Simazine	*Diazinon*	*Isodrin*
		Ipazine		*Benomyl*
		Alachlor		*Dieldrin*
		Ametryne		*Chloroneb*
		Propazine		Paraquat
		Trietazine		Trifluralin
				Benefin
				Heptachlor
				Endrin
				Aldrin
				Chlordane
				Toxaphene
				DDT

*From data of Gray and Weirlich (64), C. I. Harris (77), Helling (81), Koren et al. (101), Nash and Woolson (133), and many other references.

**Class 5 compounds (very mobile) to Class 1 compounds (immobile); in each class pesticides are ranked in estimated decreasing order of mobility.

+Names of herbicides are set in normal type; insecticides, fungicides, and acaricides in *italics*.

persistence of the compound in the soil. Available information indicates that pesticide contamination of groundwater under normal agricultural usage is minimal. However, a need exists for more information on the extent of pesticide pollution as pesticides are known to enter underlying water through cracks and through sand and aquifers. The fate of pesticides in subsurface environments and movement of contaminated groundwater to aquatic systems is absolutely unknown (155).

3.5.3 Pesticide Movement by Surface Runoff

Pesticides present on agricultural land may be transported through surface runoff either in solution and/or as adsorbed molecules on suspended eroded soil particles. Since surface soils are susceptible to erosion, pesticides retained in the plow-depth layer are potentially transportable by surface drainage. The rate and magnitude of pesticide loss by runoff depends principally on soil properties, nature of the pesticide, and climatic factors such as frequency, intensity, and duration of rainfall (9, 22, 49, 50, 84, 138). Modifying factors include topography, vegetative cover, rate of application, time between application and first rainfall, and size of the watershed (12, 15, 24, 73, 199). The movement and persistence of pesticides in the soil affect the amount of residues present in the surface soil as well as that in runoff. Factors affecting the movement and persistence of pesticides in soil have been reviewed (10, 45, 82, 95).

Surface runoff from agricultural watersheds has been implicated as the major avenue of pesticide input to the Great Lakes and other aquatic systems (35, 103, 131, 139, 148, 204), especially the organochlorine insecticides. As yet, no comprehensive effort has been made to determine the contribution of agricultural runoff to the total pesticide burden of the environment. Investigations conducted in small treated watersheds demonstrated the lateral movement of pesticides, and data from several studies are summarized in Table 12 (144). Table 12 reveals that losses of most organochlorine insecticides relative to the amount applied are low even for surface-applied insecticides. Once the soils are contaminated with the more persistent organochlorine insecticides like DDT, these

TABLE 12

Runoff losses of agriculturally applied pesticides under various conditions. (Adapted from Pionke and Chesters (144).)

Pesticides and literature reference ( )	Amount applied, kg/ha	Type of application**	Crop	Soil texture	Avg slope, %	Plot size	Duration of experiment, months	Amount of runoff, cm	Pesticide loss in runoff, % of application	Range of pesticide loss in runoff increments, ug/g	Location
Aldrin (72)	1.5	Ss	Cultivated	sil	1-2	3x1.2 m	++	4.3¶	5.2	##	KY
Dieldrin (30)	5.6	Inc	Primarily corn	sil	14	1.09 ha	26	3.1	0.07	0.4-4.1	Coshocton, OH
Dieldrin (30)	5.6	Ioc	Primarily corn	sil	14	0.68 ha	8	3.1	0.007 2.2	(1.9-20 ug/l water) (1.6-14 ug/g sediment)	Coshocton, OH
Dieldrin (72)	1.5	Ss	Cultivated	sil	1-2	3x1.2 m	++	4.3¶	4.7	##	KY
DDT (72)	1.5	Ss	Cultivated	sil	1-2	3x1.2 m	++	4.3¶	6.3	##	KY
DDT (49)	0.73*	Ss***	Cont. potatoes	gl	8	3.7x21 m	24	9.7	1.6	7.0-8.3	Presque Isle, ME
DDT (49)	0.73*	Ss***	Rotation potatoes	gl	8	3.7x21 m	24	7.9	1.0	Trace-67	Presque Isle, ME
DDT (49)	0.73*	Ss***	Oats	gl	8	3.7x21 m	12	3.6	0.01	Trace-22	Presque Isle, ME
Endosulfan (49)	1.0	Ss***	Cont. potatoes	gl	8	3.7x21 m	12	5.1	0.35	1.0-19	Presque Isle, ME
Endosulfan (49)	1.0	Ss***	Rotation potatoes	gl	8	3.7x21 m	12	4.8	0.25	Trace-18	Presque Isle, ME
Endosulfan (49)	0.72*	Ss***	Oats	gl	8	3.7x21 m	12	3.6	0.01	Trace-3	Presque Isle, ME
Endrin (49)	1.5	Ss***	Cont. potatoes	gl	8	3.7x21 m	24	9.7	0.9	1.0-49	Presque Isle, ME
Endrin (49)	1.5*	Ss***	Rotation potatoes	gl	8	3.7x21 m	24	7.9	0.6	Trace-48	Presque Isle, ME
Endrin (199)	0.337	Sg	Sugarcane	scl	0.2	7.3x61 m	10	1.2§	0.1	<0.01-2.73	Baton Rouge, LA
Endrin (199)	0.337	Sg	Sugarcane	scl	0.2	7.3x61 m	5	0.3	0.2	0.15-5.02	Baton Rouge, LA
Methoxychlor (46)	22.4	Sl	Grass	sil	-	0.07 m²†	14	27	0.04	0.1-8.8	Coshocton, OH
Atrazine (196)	3.0	Ss	Fallow	sl	6.5	1.8x10.7 m	+++	3.9	5.7-7.3	(100-10,340 ug/l water) (0.5-88 ug/g sediment)	Watkinsville, GA
Atrazine (152)¶¶	3.0	Ss	Corn surf. contour.	sil	10-15	0.8-1.5 ha	1	0.4	15	(1,170-4,910 ug/l water) (1.77-7.35 ug/g sed.)	Castana, IA
Atrazine (152)	3.0	Ss	Corn ridged contour.	sil	10-15	0.8-1.5 ha	1	1.5	2.2	--	Castana, IA
Propachlor (152)	6.0	Ss	Corn surf. contour.	sil	10-15	0.8-1.5 ha	1	1.0	2.6	--	Castana, IA
Dicamba (181)	1.0-2.0	Ss	Fallow-sod	cl	3-8	3.05x3.05 m	4	<0.2¶	2	0-4,800	College Station, TX
2,4-D-isooctyl (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	+++	10.2¶	40	1,380	Watkinsville, GA
2,4-D-butyl ether (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	+++	10.2¶	35	640	Watkinsville, GA
2,4-D-amine (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	1 hr	9.4¶	5	140	Watkinsville, GA
2,4-D-amine (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	48 hr	8.9¶	4	110	Watkinsville, GA
2,4-D-amine (12)	2.2	Ss	Cultivated	sl	7	3.6x10.7 m	96 hr	8.1¶	5	180	Watkinsville, GA
Picloram (181)	1.0-2.0	Ss	Fallow-sod	cl	3-8	3.05x3.05 m	4	<0.2¶	2.8-5.0	15-5,060	College Station, TX
2,4,5-T (46)	11.2	Sl	Grass	sil	-	0.07 m²†	14	27	0.05	1-380	Coshocton, OH
2,4,5-T (181)	1.0-2.0	Ss	Fallow-sod	cl	3-8	3.05x3.05 m	4	<0.2¶	2	7-3,300	College Station, TX

*Includes residual insecticide present

**Ss = surface spray; Inc. = incorporated 7.5 cm; Sg = surface, granular; Sl = surface, liquid

***Mixed by subsequent cultivation if used on row crops.

†Lysimeter study

††One rainfall occurrence

†††Application 1 to 96 hr before simulated rainfall

¶Generated by simulated rainfall

§Generated, in part, by irrigation

##Combined sediment-water allowed to equilibrate before analysis.

¶¶Addition to Pionke and Chesters (1973) data.

NOTE: kg/ha = 0.892 lb/A; m = 3.282 ft; ha = 2.471 A; m^2 = 10.771 ft^2; cm = 0.394 in; ug/g = ppm.

compounds will reside in the soil for an indefinite period of time and are capable of being carried from one season to the next (84). Thus, the possibility exists that persistent pesticides associated with the soil are subjected continually to runoff, thereby providing a steady, low-level residue to aquatic systems.

Generally, investigations show that pesticide concentrations in runoff are many times greater in the period immediately following application than at later times (12, 22, 30, 46, 49, 152, 181, 199). This period may last for as much as 4 months but is time-dependent. Levels at later periods would depend mainly on the pesticide reservoir residing in the soil that is available for transport. Processes such as volatilization, adsorption, degradation, photodecomposition, and leaching will determine, to a large extent, the level of residues remaining at a particular time.

Periods of accelerated loss following application are observed for several herbicides (Table 12). Rainfall or irrigation occurring immediately following application caused large losses of weakly adsorbed compounds (12, 181, 196) and accelerated loss of the more strongly adsorbed compounds (30, 49, 72, 84, 199). Losses of 2,4-D varied according to the formulation and decreased with increase in solubility. This suggests that downward movement of more soluble compounds into the soil protects them from runoff.

In most treated farmlands, loss following application appears to be affected not only by the availability of runoff-causing rainfall but also on the elapsed time between application and rainfall. Some studies show that concentration of endrin and aldrin in runoff decreased by a factor greater than two where rainfall was delayed 3 to 7 days (72, 199). The time interval from application to the first irrigation affected the amount of picloram removed with surface water or leached into the soil profile of semi-arid rangelands (157). The lower levels associated with delayed rainfall or irrigation is probably attributable to reduced quantities of residues resulting from degradation, volatilization, and photodecomposition. This is of greater importance with the more volatile and less persistent pesticides. Persistent pesticides tend to accumulate in the top soil making them accessible for transportation during periods of runoff.

It is evident that pesticide concentrations are generally higher in the sediment than in the runoff water (24, 30, 72, 196). This relative distribution between the two phases expressed as ratios of pesticide concentrations in runoff water to that adsorbed to the suspended sediment ranged from 1:2 to 1:1000 for most organochlorine insecticides (30, 72, 150, 168) and 1:10 to 1:20 for triazines (74, 196). Although concentrations were higher in the sediment, greater total losses were associated with the runoff water due to its greater volume. This emphasizes the importance of reducing erosion by sound soil management and conservation practices to minimize the pesticidal load of the runoff.

The concentrations and losses reported relate to runoff as it immediately leaves a specific watershed area. Under actual field conditions, agricultural watersheds are seldom treated completely. Studies have shown that losses of diuron, picloram, 2,4-D, and 2,4,5-T diminished as a function of distance from the treated field (50, 117, 181). All herbicide concentrations dropped below the limit of detection within a few hundred meters below the sprayed area (50). These data suggest that runoff originating from treated areas is diluted by runoff arising from untreated areas. Furthermore, uncontaminated suspended particles may adsorb the pesticide present in the water and, together with the originally contaminated sediment, become partially redeposited across the path of the runoff. The amount of a pesticide entering tributaries and lakes depends upon the reactions it undergoes during its travel over land.

Limited studies have been conducted for monitoring pesticide movement over large agricultural watersheds. In such investigations the drainage streams and/or near-shores of the lakes were sampled usually to determine pesticidal pollution from surrounding drainage areas. In Lubbock County, Texas, 18 rural lakes draining extensively-farmed areas were sampled on a routine basis following runoff-producing rainfall for a period of 18 months (195). No measurable concentrations of any of the herbicides or insecticides used commonly in the area were detected in the water samples. Dieldrin, aldrin, and DDT were the only insecticides found in the lake sediments, with dieldrin being present in almost 80% of the samples. The

sediment samples contained no detectable herbicides.

Seasonal variations in residues of organochlorine pesticides in the water of the Utah Lake drainage system were determined in 1970-71 (25). Definite surges of pesticide [1 µg/l (1 ppb) or more] enter Utah Lake three times a year, i.e., early spring, late spring, and fall, generally corresponding to the application times of pesticides by farmers in the area. The pesticides involved were mainly aldrin and BHC in early spring; heptachlor including heptachlor epoxide and methoxychlor in the late spring; and aldrin, heptachlor and methoxychlor in the late fall.

Runoff losses of DDT and dieldrin occurring in a 40,500 ha (100,000 acres) watershed draining the tobacco belt of southern Ontario to Lake Erie were determined by Frank et al. (58). It was calculated that 10 kg (22 lb) of DDT and 0.6 kg (1.3 lb) of dieldrin reached Lake Erie from the watersheds in runoff water or on suspended material. This represented 0.003% of DDT and 0.004% of the dieldrin resident in the watershed soil which were 325,000 kg (716,300 lb) and 14,500 kg (32,000 lb), respectively. As water and sediment entered Long Point Bay and moved into Lake Erie, insecticide residues became considerably diluted.

In October 1968, chlordane and dieldrin were applied on 1,710 ha (4,225 acres) of land bordering Lake Michigan in southeastern Michigan (Chikaming Township, Berrien County) to control Japanese beetle infestation. The aerial application was made using 2,824 kg (6,225 lb) of technical dieldrin and 5,043 kg (11,115 lb) of technical chlordane. The impact of this application to Lake Michigan was monitored by residue measurements in water, sediment, and mussel tissue before and after treatment at two control and four test stream stations (204). Prior to treatment, mean concentrations of chlordane were <0.2 µg/l (<0.2 ppb) in the water, <50 ng/g (<50 ppb) in the sediment, and 34 ng/g (34 ppb) in mussel tissue. The mean dieldrin concentrations before treatment were <0.02 µg/l (<0.02 ppb) in water, <5 ng/g (<5 ppb) in sediment, and 7.83 ng/g (7.83 ppb) in mussel tissue. Following treatment, the chlordane concentrations in water, sediment and mussel tissue reached individual station highs of 3.4 µg/l (3.4 ppb), 22,000 ng/g (22 ppm), and 7,530 ng/g (7.53 ppm), respectively; and dieldrin

concentrations reached 2 µg/l (2 ppb), 2,000 ng/g (2 ppm) and 1,137 ng/g (1.137 ppm), respectively.

Movement of pesticides from treated fields to stream waters occurred in the first 3 months after application as shown for dieldrin (Fig. 3). Based on the discharge flow to the streams an estimated 5.1 kg (11.3 lb) of dieldrin was contributed to Lake Michigan by stream water in the 21 months following treatment; this represented 0.18% of the total application of 2,824 kg (6,225 lb) dieldrin.

Stream sediments continued to show low, but detectable concentrations of chlordane 12 months after treatment. No further monitoring was done after this period. Dieldrin was still present in the sediment 21 months after application at concentrations of 60 to 250 ng/g (60 to 250 ppb). Chlordane and dieldrin concentrations in the stream sediments were high during periods of high flow. This is indicative of the strong affinity of these pesticides to the eroded soil particles and/or to suspended sediments in the streams. Probably bedload, allochthonous, and autochthonous organic materials play significant roles in the transport and accumulation of insecticides in the Great Lakes.

To determine whether North Shore streams in the Minnesota drainage basin of Lake Superior are important contributors of pesticides, samples of water and clams collected from 24 streams were analyzed for *p,p'*-DDT, *o,p'*-DDT and DDE within an 8-month period starting in May, 1972 (127). Under conditions of normal flow, very little insecticide contamination was being contributed to Lake Superior by those streams. Highest values [20 to 21.8 µg/l (20 to 21.8 ppb) *p,p'*-DDT] occurred in water samples from Lester and Lower St. Louis Rivers following a heavy runoff in October. Ordinarily, median levels in these rivers were 0.03 and 0.02 µg/l (0.03 and 0.02 ppb), respectively. The median value for all other streams was 0.01 µg/l (0.01 ppb). Native clams from the St. Louis River had accumulated appreciable amounts of *p,p'*-DDT ranging from 93 to 1,630 µg/l (93 to 1,630 ppb) indicating that this river carries an appreciable load of DDT on either an intermittent high level during heavy rains or a constant low level basis. Although this study did not isolate the sources of contami-

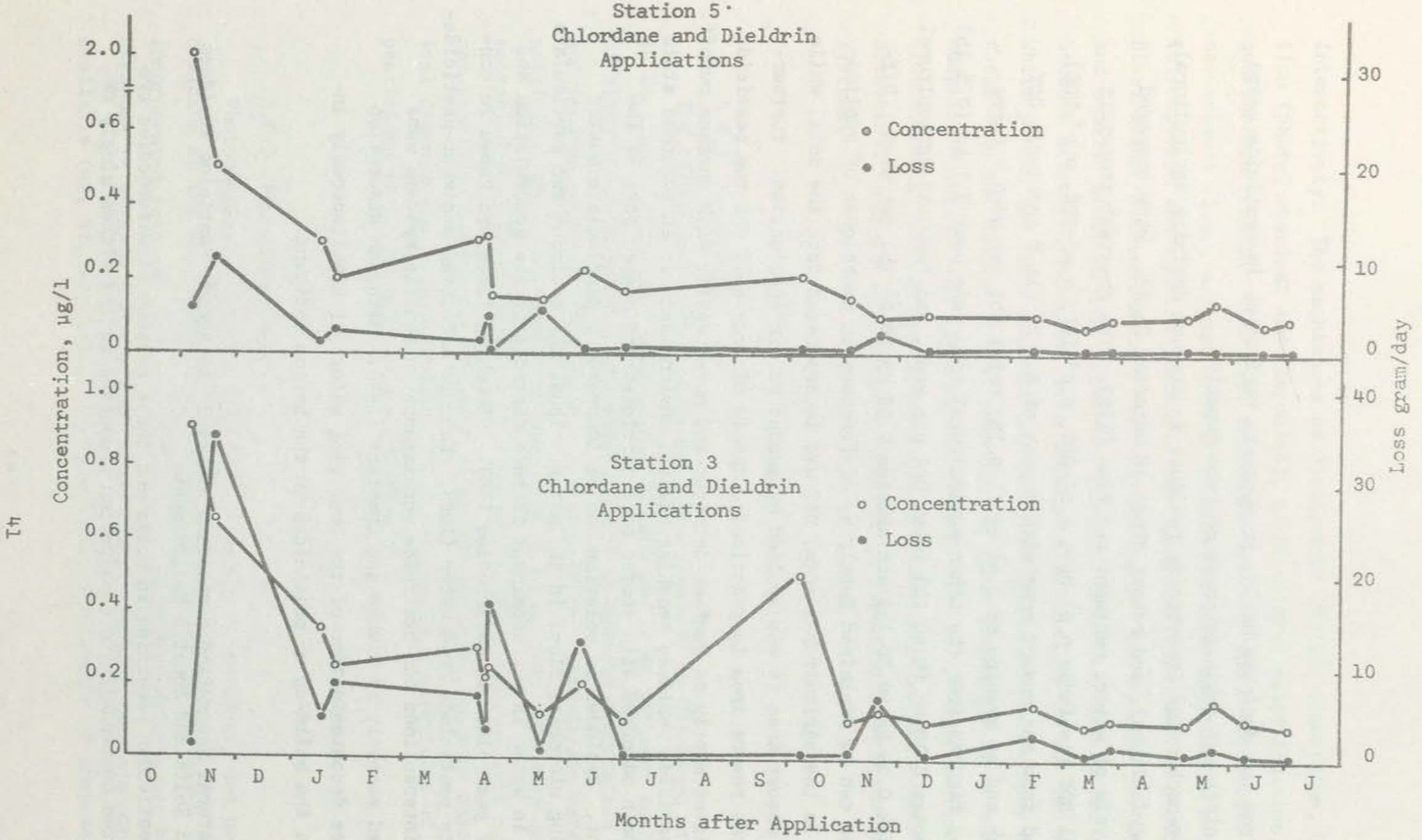

Fig. 3. Concentrations and losses of dieldrin to Berrien County (Michigan) streams, October 1968-July 1970. Modified from Willson et al. (204). µg/l=ppt, gram/day = 0.035 ounces/day.

nants from the drainage basin, it suggests variations in pesticide usage in the different drainage areas of Lake Superior.

Organochlorine insecticide residues in streams draining agricultural, urban-agricultural, and resort areas of Ontario, Canada, were compared by analysis of water, sediment and fish (125). The greatest transport of total DDT (includes *p,p'*-DDT, *o,p'*-DDT, *p,p'*-TDE, *o,p'*-TDE, *p,p'*-DDE) occurred from the resort area with a peak of 5.4 kg (11.8 lb) total DDT per week and an average of 0.86 kg (1.9 lb) total DDT per week. Corresponding figures from the urban-agricultural area were peak 1.1 kg (2.5 lb) and average 0.18 kg (0.40 lb) total DDT per week, and from the agricultural area peak 0.23 kg (0.50 lb) and average 0.09 kg (0.20 lb) per week. Differences can be explained partly by differences in techniques of application. In the agricultural area, DDT was incorporated into the soil, while in the resort area it was applied by ground or air application. Furthermore, the resort area is practically devoid of true soil and the pesticide accumulated mostly on surface detritus and moved easily with surface runoff.

Pesticide residues reaching aquatic environments in an adsorbed state can remain adsorbed with eroded soil particles and become part of the sediment. Residues in solution can be adsorbed by particulate matter including microbial flocs in the water. Thus, the sediment and particulate matter in water is a predominant feature controlling the accumulation and fate of pesticides in waterbodies (36). Microbial flocs are known to concentrate pesticides from water (106). Factors affecting sediment-pesticide-water interactions and pesticide persistence in aquatic systems were reviewed recently by Pionke and Chesters (144). Although adsorption enhances decontamination of the overlying water, it simultaneously increases the build-up of pesticides in the bottom sediment.

3.6 Natural Renovation Mechanisms Available to Remove Pesticide Residues in Soils and Aquatic Environments

Pesticides remaining in soils and those present in waterbodies are subjected continually to dissipation processes acting either singly or

interactively. The mechanisms of dissipation include adsorption, degradation (photo, chemical and microbial), plant uptake, volatilization, leaching, and surface runoff. The last three are means by which pesticides move from one segment (i.e., agricultural watersheds) of the ecosystem to another (atmospheric and aquatic systems), and these are discussed in Section 3.5. Dissipation processes except leaching and runoff are similar for both soil and aquatic environments. However, rates of pesticide dissipation may vary because of the differences in environmental conditions in the two systems. While pesticides have inherent but varying degrees of persistency because of their chemical nature, their longevity in these environments is modified by edaphic, climatic, and limnological factors (144). In this section, pesticide losses resulting from adsorption, degradation, and plant uptake are presented, followed by a summary of their persistence in soil and aquatic systems.

3.6.1 Adsorption

The effect of adsorption on pesticide movement is discussed at length in Sections 3.5.1, 3.5.2 and 3.5.3. Adsorption is an indirect mechanism of dissipation from the environment because it affects degradation rates and reduces pesticide bioactivity and volatility. Degradation rates of pesticides may be decreased or increased by adsorption depending upon the mechanism by which the compound is degraded. Adsorption by soil or sediment can stabilize pesticides against biodegradation while, in some cases, chemical degradation is accelerated. Furthermore, the availability of adsorbed pesticides to plants may be reduced markedly.

3.6.2 Photodecomposition

Photochemically-induced degradation occurs at surfaces, and pesticides residing at the soil surface are more subject to photodegradation than those incorporated into the soil. Additionally, photodecomposition has been observed in aqueous environments. Reviews on pesticide photodecomposition are available (40, 41, 42, 145). The practical significance of photodegradation

as a means of pesticide removal in soil and aquatic systems has not been determined quantitatively because of the difficulty of interpolating data gathered in the laboratory to field conditions. In natural systems, photochemical reactions occur simultaneously with adsorption and microbial decomposition, but in most cases the photodegradation reactions are comparatively slow. However, the possibility exists that photodecomposition contributes to the airborne dissipation of some pesticides.

Photochemical degradation has been demonstrated for many pesticides including a number of organochlorine insecticides. DDT was found to degrade slowly in sunlight to DDE, TDE (DDD) and other products (146). Aldrin and dieldrin, upon exposure to sunlight, were converted to photoaldrin and photodieldrin, respectively (153). Photoaldrin was approximately twice as toxic as the parent compound to insects and mice. Other related insecticides photolyzed by sunlight include chlordane, endrin, heptachlor, isodrin and methoxychlor.

3.6.3 Plant Uptake

The use of trap plants to remove pesticides from soil and water systems has been suggested (55). This potential environmental cleanup method was based on the fact that many terrestrial and aquatic plants are capable of absorbing and translocating pesticides (45, 96), followed by possible detoxification of the compound to less active components.

Several terrestrial crops were found to absorb chlorinated insecticides (17, 110, 113, 132, 135), and evidence of metabolic breakdown was indicated for DDT, heptachlor, endrin, γ-BHC and aldrin. Corn, which is resistant to the *s*-triazine herbicides--atrazine and simazine--absorbed these herbicides from soils and metabolized them to nonphytotoxic compounds (129). Dissipation of atrazine from soils through uptake by corn, sorghum and johnsongrass has also been studied (169).

Many species of algae and aquatic plants were able to remove low concentrations of herbicides, namely, 2,4-D, amitrole, atrazine, dicamba, dichlobenil and diphenamid, from water (21). Once absorbed, most of the

pesticides were metabolized by various aquatic plants and algae. Metabolism was rapid, particularly with the algae. In some instances, portions of the metabolites and undegraded herbicides were released back to the water. In another investigation it was shown that algae were able to concentrate pesticides by a factor of severalfold and were generally more resistant to pesticide toxicity than higher members of the food chain. This suggests that phytoplankton and aquatic weeds can remove effectively the low-level concentrations of pesticides normally encountered in aquatic systems and metabolize them to less active compounds.

Sorption of herbicides by weeds results frequently in the death of the plant. If degradation of the herbicide is slow, the sorbed herbicide may be returned to the soil or lake sediment where the dead weeds undergo decomposition. For example, in an aquatic weed control program, using paraquat and diquat, the herbicides were not detected in the sediment until the dead weeds had settled to the bottom sediment and had been subjected to decomposition processes (57). Upon decay of the dead weeds the herbicides were either released and adsorbed by the sediment or remained bound with the settled organic material. In addition, certain pesticides may be absorbed and translocated without degrading as shown in the case of dieldrin uptake by corn (19), or simply may be converted to another active compound as was found for enzymatic conversion of aldrin to dieldrin in peas (110). Plant-absorbed pesticides which are resistant to metabolic breakdown within the plant either become part of the food chain or are returned to the soil or sediment.

Probably the extent of pesticide detoxification by plants is small, as uptake is limited by spatial availability of the pesticides and by the sorption capacity of the plant. However, a great need exists to better understand the mechanism of pesticide dissipation through plant uptake. Included in these investigations must be a search for terrestrial and aquatic plants that are efficient in detoxifying a wide spectrum of pesticides absorbed by them.

3.6.4 Chemical Degradation

Nonbiological processes of pesticide breakdown in soils and sediments have long been recognized. Chemical reactions of pesticides may occur independently of soil or they may be soil-catalyzed. Extensive coverage of this topic is provided in the comprehensive review by Helling et al. (82).

The chemical hydrolysis of organochlorine insecticides has not received as much attention as that of the organophosphorous insecticides and herbicides. Evidence for the chemical decompositions of chlordane, toxaphene, heptachlor, DDT, dieldrin and endrin have been reported in dry, acidic clay diluents used in insecticidal formulations (53). Chemical conversion of endrin to two isomers--ketone and aldehyde--was observed in dry, acidic soils but no conversion occurred in the presence of moisture (53, 134). DDT conversion to DDE has been shown to occur in moist and dry soils ranging in texture from sandy loam to clay under a wide range of pH values (136). Increasing the soil pH by liming and additions of Fe_2O_3 and Al_2O_3 to the soil failed to enhance conversion of DDT to DDE. Data indicate that the predominant mechanism of conversion of DDT in moist soil is by a microbial pathway, while chemical degradation is predominant in dry soil. Similar findings have been reported for heptachlor (23). Conversion of heptachlor to 1-hydroxychlordene was rapid in a series of dry soils of low organic matter content. The reaction did not proceed in soils of high organic matter content.

Results of these investigations indicate that chemical breakdown may play a significant role in the dissipation of soil-adsorbed organochlorine insecticides during dry periods. However, much more information is required on the rates of these conversions and on the environmental conditions which promote this type of organochlorine insecticide dissipation from the environment.

Several organophosphorous insecticides and herbicides have been shown to degrade rapidly by chemical hydrolysis. The process is catalyzed by adsorption at soil colloid surfaces and follows first-order kinetics (7, 8, 98, 99, 100). In soil systems, chemical hydrolysis of malathion

was rapid and was almost completed before the termination of the lag phase necessary for promotion of the biological degradation mechanism for malathion (100). Alkaline conditions in the soil enhanced degradation of malathion by chemical hydrolysis (186). Other organophosphorous pesticides found to degrade chemically are diazinon (99, 164), phorate (61), dichlorvos (62), ciodrin (98), and imidan (122). The longer-lived organophosphorous pesticides, including methyl parathion, parathion (62, 63, 165), dimethoate, zinophos, and dursban (62) are degraded primarily by microbial mechanisms.

Chemical hydrolysis of the 2-chloro-*s*-triazine herbicides in soils and sediments has been reported (7, 8, 77, 171). The hydrolysis of atrazine to nonphytotoxic hydroxyatrazine is enhanced by atrazine adsorption possibly to carboxyl groups present on the organic components of soils and sediments (7).

Although certain pesticides are able to undergo rapid chemical degradation, formation of degradation products as persistent, or more so, as the parent compound may occur. For instance, among the degradation products of diazinon is diethyl thiophosphoric acid which is more stable in soils than the parent compound (100). A gap still exists in the understanding of the chemical mechanisms of degradation of the more persistent pesticides, particularly the organochlorine insecticides, as well as the fate and toxicity of the resulting degradation products.

3.6.5 Microbial Degradation

Microbial metabolism is considered to be the major pathway of degradation for many pesticides in soils and sediments. The efficiency of this pathway depends upon such environmental factors as temperature, moisture content, organic matter content, aeration, pH, and pesticide concentration. Although intensive studies have been made on the mechanisms by which microorganisms degrade pesticides, the processes are not understood clearly.

In general, the organochlorine insecticides are the most resistant pesticides to microbial attack. Degradation products encountered frequently

are compounds which retain insecticidal properties. For example, partial degradation of DDT results in the formation of TDE (DDD) and DDE. TDE (DDD) and DDE are stable in soil and aquatic systems, and their metabolic fate in these environments remains relatively unknown. Several organochlorine insecticides although relatively persistent may degrade at a potentially significant rate in soils. Heptachlor (126), lindane (209), and endrin (23) have been shown to degrade in soils to compounds of reduced insecticidal activity. Conversion of heptachlor to the less toxic metabolite 1-hydroxychlordene also has been reported (126). The oxygen status of soils and aquatic systems has pronounced effects on the microbial breakdown of many organochlorine insecticides. In soils, DDT is rapidly converted to TDE (DDD) under anaerobic conditions and very slowly to DDE under aerobic conditions (68, 69, 95). The addition of such readily available energy sources as alfalfa meal or sucrose promoted the anaerobic disappearance of DDT from soils (26, 69). However, in aerobic soils DDT remained stable despite the addition of alfalfa meal. These observations suggest that flooding DDT-contaminated soils may accelerate decontamination. Conditions conducive to anaerobiosis, like thermal stratification of lakes, may also play an important role in the disappearance of DDT accumulated in bottom sediments. In raw water containing some colloidal material and maintained under aerobic conditions, heptachlor and endosulfan were degraded rapidly while lindane, heptachlor epoxide, dieldrin, DDT, TDE (DDD), DDE and endrin were not degraded (47). Endrin, *p,p'*-DDT, lindane, aldrin, and heptachlor were degraded in anaerobic digested wastewater sludge to various derivatives (83). Although conditions in the sludge are not characteristic of most lakes, the data emphasize the role of oxygen deficiency in the metabolic fate of many persistent pesticides.

The conversion of *p,p'*-DDT to *p,p'*-DDD has been observed in flooded anaerobic soils (34, 69) and in oxygen-deficient lake water (128). The time required to convert 50% of the applied DDT to TDE (DDD) was 24 hr in the lake water and 8 weeks in the flooded soils. However, TDE (DDD) formed in flooded soils seemed to resist further degradation (34, 71). Anaerobic decomposition of DDT to TDE (DDD) has been observed in lake

sediments, but the degradation product TDE (DDD) appears to be relatively stable in lake sediments (120, 140). The rapid degradation of lindane has been reported in flooded rice soils (208), lake muds (144), and in simulated lake impoundments (137). It was estimated in the simulated lake impoundment that only 15% of the lindane was degraded in the aerobic system in 90 days while degradation was almost complete in the anaerobic system in the same time period (137). The major intermediate product volatilized rapidly. In addition, methoxychlor and heptachlor were also found to degrade in submerged soils, and the rate of loss was accelerated in the presence of high organic matter content (34).

As stated earlier, many organophosphorous insecticides are hydrolyzed rapidly by nonmicrobial means. However, for the more persistent types, microbial breakdown may predominate. Thus, conditions retarding microbial activity may result in enhanced persistence of the organophosphorous insecticides.

Soil and lake sediment microorganisms have been implicated in the degradation of several organophosphorous insecticides including diazinon (162), phorate (114), malathion (119, 186), and parathion (63, 111, 165). The degradation of diazinon (164) and parathion (165) was considerably increased in flooded soils. Apparently, the presence of molecular oxygen retarded microbial degradation although this was not observed for parathion in lake sediments (63). The degradation of parathion to aminoparathion was found to be greatly accelerated by microbial activity, and the rate was similar under aerobic and anaerobic conditions. These results imply that in sediments aerobic and anaerobic organisms are able to degrade parathion. Adsorption of some organophosphorous insecticides may increase their persistence in aquatic systems. For example, dursban was shown to persist for much longer periods of time in aquatic systems of high colloidal organic matter content due to pesticide adsorption (156).

Microbial degradation of organochlorine and organophosphorous insecticides in aquatic environments was reviewed recently (141, 163). In a review of the degradation of the newer vinyl phosphate insecticides, such as phosphamidon, chlorfenvinphos, and mevinphos, it has been shown that

the half-life of these pesticides in soils ranges from 1 week to 7 months (20). Chlorfenvinphos appeared to be the most resistant to biological decomposition.

Despite the increasing use of the carbamate insecticides, investigation of their degradation has not been as extensive as for other insecticidal groups (158). Carbaryl, the most widely used carbamate, was found to be readily degraded by microorganisms in soils (116) and natural waters (6). Some members of the group, such as dimetilan and pyrolan, tended to be more biologically stable in natural waters possibly due to their strong adsorption on suspended clay minerals (6).

Most of the agriculturally-used herbidides are degraded primarily by microorganisms. A great deal of study has been devoted to their degradation and exhaustive reviews have been compiled (92). Many of the herbicide groups, including the thiocarbamate (93, 172), phenylcarbamate (91), substituted urea (91, 94), and the commonly used chlorinated aliphatic acids (54) and phenoxyalkanoic acids (115), are microbiologically degraded in soils in relatively short periods. Compared to organochlorine insecticides, phenoxyalkanoic acids, such as 2,4,5-T and particularly 2,4-D, degrade rapidly in soils and sediment-water systems (55, 160). Anaerobic conditions tend to retard 2,4-D metabolism, indicating the importance of oxygen in the metabolism of this compound in the environment (104). Several of the benzoic and phenylacetic acid herbicides are more persistent. For instance, dicamba, 2,3,6-TBA, fenac and methoxyfenac are comparatively more resistant to microbial attack than chloramben (166) which is degradable readily by microbial processes (198). Picloram and trifluralin are considered to be persistent herbicides (39). In a long-term field experiment picloram residues were detectable 9 to 15 months after application (123). Propanil, an anilide herbicide, was found to be microbially-degraded primarily to chloranilines (13). However, the metabolites are adsorbed strongly by soil organic matter and may exist in the soil for several years. Some of the easily degradable herbicides may persist longer in aquatic than in terrestrial systems. The substituted urea herbicides, monuron and neburon, persisted in excess of 2 years in simulated ponds

containing amorphous materials (56) while CIPC--a carbamate herbicide--persisted under anaerobic conditions induced in an artificial impoundment (161).

The *s*-triazine herbicides, as indicated earlier, can be extensively degraded chemically in soils or sediments, but they are also subject to microbiological metabolism (91). Atrazine, the most widely used *s*-triazine, is quite stable compared with most of the herbicides and can persist in soil for more than one growing season (27, 104). Rates of atrazine degradation are influenced greatly by the spatial location of the herbicide in the soil profile. Atrazine present in the plow layer persisted for 5 months while atrazine at 40- and 90-cm (15.7- and 35.4-in) depths persisted for 17 months and 41 months, respectively (104), which suggests the importance of management techniques in minimizing atrazine leaching.

Adsorption of pesticides by clay minerals may protect them from microbial attack. The dipyridyl herbicides--paraquat and diquat--are shown to be metabolized by soil microorganisms (59), but once adsorbed by clay minerals they become extremely resistant to microbial decomposition (191). In aerobic and anaerobic aquatic systems, diquat was not degraded within 6 months (170).

Figure 4 summarizes the persistence of 11 major pesticide groups (95). The persistence values represent a 75 to 100% loss of the pesticide. As indicated elsewhere in this section, persistence of pesticides in aquatic environments may differ from that in terrestrial soils, but more investigations are needed to make generalizations. Furthermore, to elucidate the metabolic products, sustained and continuing research is needed in order to better understand the environmental implications of pesticides, particularly in the aquatic segment of the ecosystem.

3.7 Probable Changes in Land Use Activities with Time

Tables 1 and 2 show that a general decrease in agricultural land area in the U.S. portion of the Great Lakes Basin will occur for the next 50 years. However, the decline is projected to be slight, at approximately 5% for cultivated cropland and 8% for pasture land. If predictions by crop

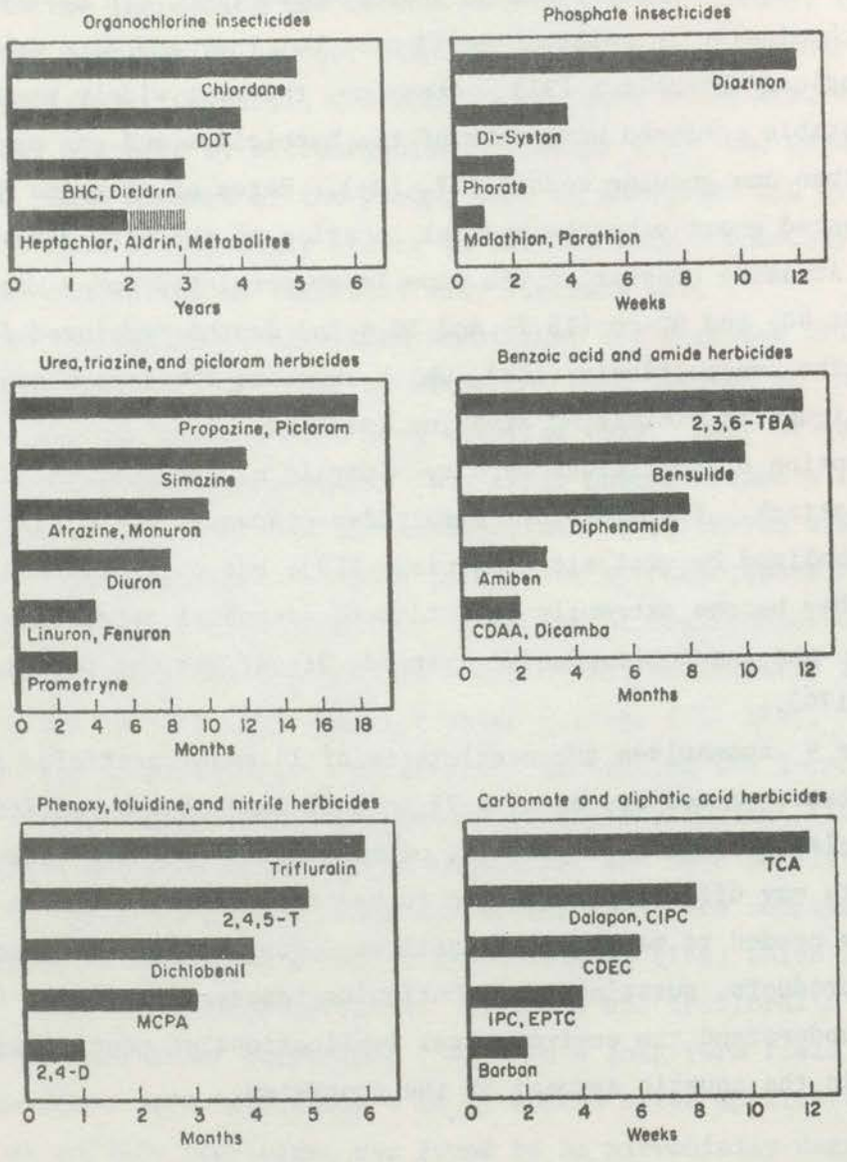

Fig. 4. Persistence of pesticides in soils. Redrawn from Kearney et al. (95).

category are considered, area for row and specialty crops will increase by 22 and 5%, respectively, as a result of projected increases in areas sown to corn, soybeans and commercial vegetables. As urbanization expands, thereby placing increased pressure on cultivated and pasture lands, some of the present idle cropland will likely be brought into cultivation. By the year 2020 it is estimated that the area of idle cropland will decrease by about 26%.

Estimated population increases in the region (65) will exert a marked influence on crop production. Since the projected area devoted to crop production is substantially unchanged, higher crop production per unit area will be needed to meet the food requirements of a rapidly-expanding population by the year 2020. Simultaneous growth of the livestock industry will probably follow the population trends, making it necessary also to increase feed production per unit area. The tremendous increase in the projected crop-yield increase per unit area is illustrated in Fig. 5. The figure suggests clearly that the area presently under cultivation will be cropped much more intensively to double or even triple the current level of production.

In order to sustain the high level of crop yields, the use of more pesticides is inevitable. Alternative methods of control for most pests are still not available. Although research on alternative methods are continuing, development of a widely-accepted and economically-feasible method may not be available in the near future. In the Great Lakes Region, pesticide usage will increase, particularly in the case of herbicides, as indicated previously in Section 3.1.2. The use of organochlorine insecticides is curtailed greatly due to the numerous regulatory restrictions placed on them. However, their use is being replaced by the use of organophosphorous and carbamate pesticides. The quantities of these compounds used for agricultural purposes are expected to increase, and an expanded research program on the stability, degradability and toxicology of the carbamate and organophosphorous insecticides is required. Furthermore, as new pesticides are introduced, a capability should be developed so that the impact of these compounds on the environment can be predicted.

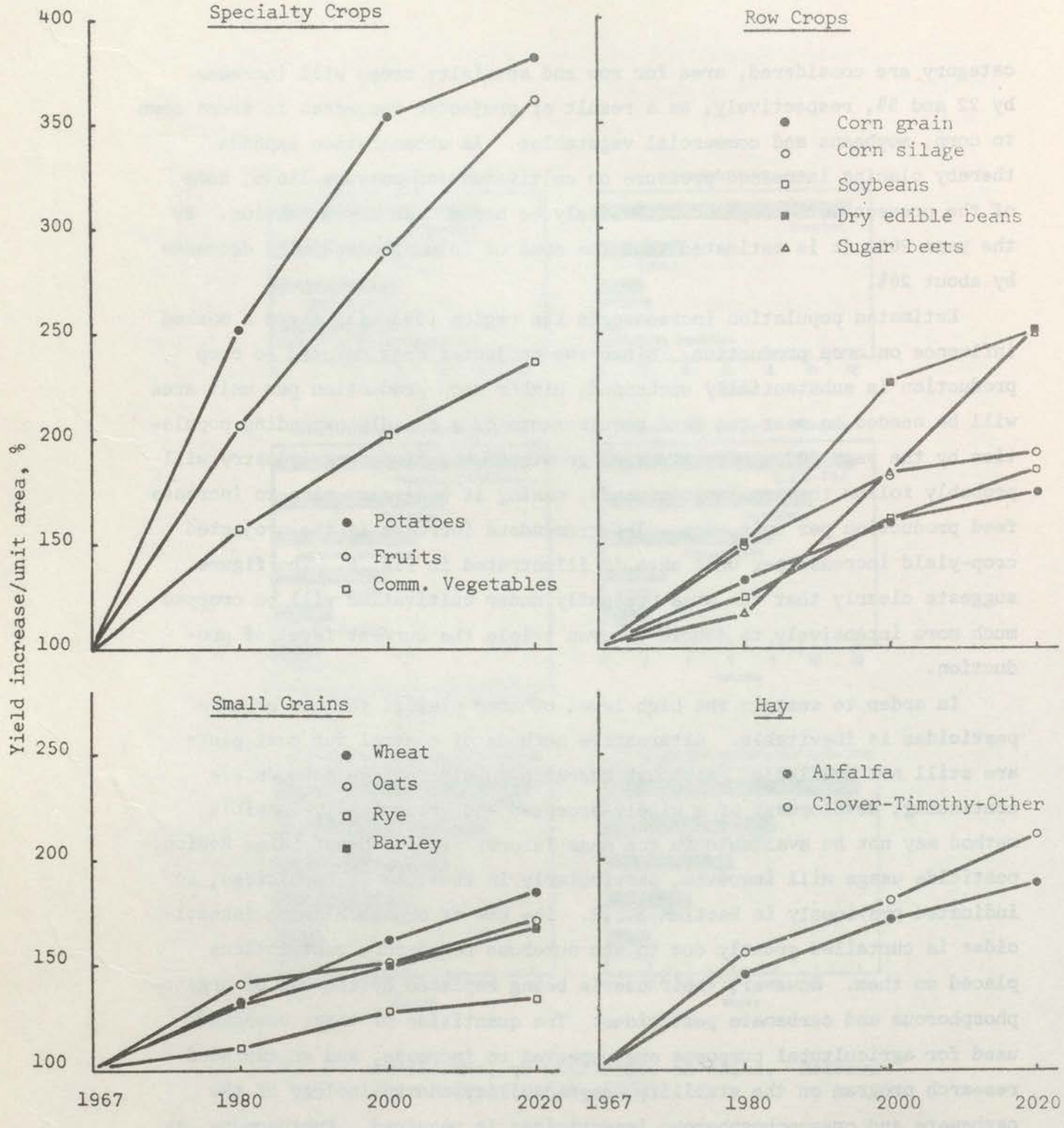

Fig. 5. Projected yield increases per unit area of major crops in the Great Lakes Region, U.S. for 1980, 2000, 2020 using 1967 as the base year. (Assembled from data in Table 1.)

3.8 Projected Seriousness of Future Pesticide Pollution Problems

3.8.1 General Level of Pesticide Contamination of Agricultural Soil and the Great Lakes

In order to assess the seriousness of pesticide pollution, it is important to know the extent of pesticide contamination of the soil and aquatic environments. However, only general indications of pesticide pollution can be made because adequate assessment is undoubtedly difficult to undertake.

Studies discussed in Section 3.5.3 indicate that the absolute amount of pesticide residues lost from agricultural watersheds through runoff is usually in the µg/l (ppb) range. Portions of the pesticides applied are associated with the soil, and the longevity of these residues is dependent upon the rates of natural renovation mechanisms and nature of the chemical. A summary of the residue content of the most commonly used organochlorine insecticides in several agricultural soils of the United States were summarized by Edwards (45). On the average, most agricultural soils contained between 1 to 4 mg/kg (1 to 4 ppm) DDT, including analogs, with the exception of orchard soils which contained substantially higher amounts, ranging from 19 to 74 mg/kg (19 to 74 ppm). Concentrations of γ-BHC, aldrin, dieldrin, chlordane, and heptachlor were generally less than 1 mg/kg (1 ppm). Pesticide residues in cropland soils for 43 states, reported by the National Soils Monitoring Program, are shown in Table 13 (197). Fruit and vegetable soils contained, on the average, the highest level of DDT of 1.92 mg/kg (1.92 ppm), which is six times higher than the national average of 0.31 mg/kg (0.31 ppm). The DDT-metabolites--DDE and TDE (DDD)--were also highest in these soils. These high levels are caused by repeated application of DDT in many types of orchards and confirms the results summarized earlier in this section. Other than orchard soils, residues found in cotton and vegetable soils were highest, while the lowest amounts were found in soils under corn and small grains. The corn, vegetable, and vegetable and fruit soils had the highest residues of dieldrin,

TABLE 13

Mean pesticide residues in mg/kg (ppm) in soil for various cropping regions in the United States, FY 1969. (Adapted from Wiersma et al. (197).)

Compound	Corn	Cotton	Cotton and General Farming	General Farming	Hay and General Farming	Irrigated Land	Small Grains	Vegetable	Vegetable and Fruit	Range of Detected Residues	No. of Samples Analyzed
Aldrin	0.05	<0.01	<0.01	0.02	<0.01	<0.01	<0.01	<0.01	0.01	0.01-3.06	1,729
Arsenic	7.44	6.72	4.88	5.35	6.42	4.77	5.70	8.75	3.27	0.25-107.45	1,726
Atrazine	0.02						<0.01			0.01-1.55	199
Carbophenothion			-	-		-				0.23	66
Chlordane	0.09	<0.01	0.01	0.01	0.03	0.03	<0.01	<0.01	0.14	0.01-6.30	1,729
2,4-D				<0.01			<0.01	-		0.01-0.03	188
DCPA	-	-	-	-	-	<0.01	-	-	-	0.54	1,729
o,p'-DDE	<0.01	0.01	<0.01	<0.01	<0.01	0.01	-	<0.01	0.01	0.01-0.20	1,729
p,p'-DDE	0.01	0.16	0.13	0.07	0.05	0.18	<0.01	0.18	0.37	0.01-6.99	1,729
o,p'-DDT	0.01	0.09	0.04	0.05	0.03	0.05	<0.01	0.07	0.06	0.01-6.29	1,729
p,p'-DDT	0.06	0.54	0.22	0.25	0.20	0.19	<0.01	0.50	0.64	0.01-35.92	1,729
DDTR	0.14	0.87	0.44	0.43	0.30	0.48	<0.01	0.81	1.92	0.01-78.36	1,729
DEF	-	-	-	<0.01	-	-	-	-	-	0.12	1,729
Diazinon		-	-			0.01				0.02-0.15	66
Dicofol	<0.01	-	-	-	<0.01	0.01	-	-	-	0.03-1.07	1,729
Dieldrin	0.05	0.01	<0.01	0.03	0.02	0.02	<0.01	0.05	0.04	0.01-1.60	1,729
Endosulfan (I)	<0.01	-	-	-	<0.01	<0.01	-	-	-	0.01-0.24	1,729
Endosulfan (II)	<0.01	-	-	-	<0.01	0.01	-	<0.01	-	0.01-0.53	1,729
Endosulfan sulfate	<0.01	-	-	-	<0.01	0.01	-	<0.01	-	0.01-0.94	1,729
Endrin	<0.01	<0.01	<0.01	<0.01	-	0.01	<0.01	0.01	0.01	0.01-0.56	1,729
Endrin aldehyde	-	-	-	-	-	-	-	-	<0.01	0.03	1,729
Endrin ketone	-	0.01	<0.01	-	-	<0.01	-	<0.01	<0.01	0.01-0.13	1,729
Ethion		-	-			<0.01				0.03	66
Ethyl parathion		-	-	<0.01	-	<0.01				0.01-3.01	66
Heptachlor	0.01	-	<0.01	<0.01	<0.01	<0.01	-	-	<0.01	0.01-0.97	1,729
Heptachlor expoxide	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01-1.08	1,729
Isodrin	<0.01	-	-	<0.01	-	-	-	-	-	0.01-0.03	1,729
Lindane	<0.01	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	-	0.01-0.35	1,729
Malathion		-	-			-				0.04-0.36	66
Methoxychlor	-	-	-	-	-	-	-	<0.01	-	0.28	1,729
PCNB	-	-	-	<0.01	-	-	-	-	-	0.69	1,729
o,p'-TDE	<0.01	<0.01	<0.01	0.01	<0.01	0.01	-	0.01	0.15	0.01-4.52	1,729
p,p'-TDE	0.05	0.07	0.04	0.04	0.01	0.04	<0.01	0.05	0.70	0.01-31.43	1,729
Toxaphene	<0.01	0.42	0.20	0.16	-	0.14	-	0.01	0.08	0.10-11.72	1,729
Trifluralin	<0.01	0.01	<0.01	<0.01	-	0.01	<0.01	<0.01	<0.01	0.01-0.25	1,729

NOTE: Blank = not analyzed; - = not detected

i.e., approximately twice as high as the national mean of 0.03 mg/kg (0.03 ppm). These data represent the residues left in the soil which may have built up due either to repeated application or high intensity of application. Thus, the soil serves as an effective reservoir of pesticide residues and metabolites which are available potentially for further transport. Although pesticide residue levels in cropland soils of some agricultural watersheds of the Great Lakes Region are well-documented and investigated, no comprehensive effort has been made to monitor pesticide residues on a continuing basis.

Residues reaching the Great Lakes are low, ranging from ng/l (ppt) to µg/l (ppb) levels. This is clearly demonstrated by the results of monitoring studies conducted on lake and tributary waters (58, 103, 109, 127, 204). Although the amounts of pesticide reaching the Great Lakes are low-level residues, contamination is continual as a result of the perpetual process of runoff and erosion occurring from adjacent treated agricultural watersheds. Furthermore, the impact of pesticides on aquatic organisms is of major concern. Fish, mussels, shrimp, invertebrates, plankton, aquatic plants, and other aquatic organims, through the process of biomagnification, concentrate the low-level residues in their tissues (18, 32, 45, 103, 127, 147, 148, 185, 204). Clearly, the hazards of pesticides in the aquatic environment lie in the biological concentration process at all trophic levels in the food chain. Effects of food chain contamination on fish and wildlife have already been well-documented and reviewed (35, 45, 103, 130, 207). Pesticide residue accumulation in the food chain may also result in hazardous exposure to humans.

Current information reveals that among the Great Lakes, Lake Michigan has the greatest pesticide problem (105, 148, 149). In view of this, most of the monitoring programs have been directed at that lake. Recently, an evaluation of DDT and dieldrin in Lake Michigan was made based on monitoring waters, biological indicators, fish and sediments for these compounds (103). Concentrations of DDT and dieldrin in lake and tributary waters, as well as in stream sediments, are presented in Tables 14, 15 and 16, respectively. The levels of DDT and dieldrin in the waters are normally

TABLE 14

Lake Michigan Open Water Pesticides--July 1969. (Adapted from The Lake Michigan Interstate Pesticide Committee of the Lake Michigan Enforcement Conference (103).)

Sample No.	Location (1/2 mile off shore)	Total DDT	Dieldrin	BHC	Estimated PCB
		ng/l			
3	Little Suamico River	<1	1.1	111	2.5
4	Pensaukee River	2.6	2.6	140	6.5
5	Oconto River	12.7	1.3	50.3	
6	Peshtigo River	<1	<1	33.0	2.8
7	Menominee River	1.1	1.5	16.5	2.5
9	Clark Lake Creek	3.5	1.5	13.7	9.0
11	Kangaroo Lake Creek	<1	1.7	8.0	Off Scale
14	Ahnapee River	15.3	1.0	23.3	50.8
15	Kewaunee River	<1	1.1	8.5	
16	East Twin River	8.8	2.2	20.0	19.5
21	Pine Creek	31.2	2.6	7.5	Off Scale
26	Pigeon River	<1	<1	11.2	2.5
30	Kinnickinnic River	5.5	2.2	8.0	9.5
31	Menomonee River	12.0	1.1	20.2	15.0
32	Oak Creek	10.8	3.3	22.8	27.2
33	Root Creek	5.9	4.5	110	5.5
34	Pike River	2.1	3.1	27.8	
35	Barnes Creek	5.0	1.9	9.0	
36	Calumet River at Calumet City	4.1	2.1	7.8	55.9
37	Burns Ditch	4.7	2.3	8.3	9.3
38	Trail Creek	3.5	2.1	1.5	6.5
39	Galien River	10.1	3.4	4.5	15.0
40	Drain at Sawyer	1.5	1.9	7.4	
42	Paw Paw River	11.1	3.0	34.3	28.3
43	Black River	1.2	1.8	1.8	2.5
48	Muskegon River at mouth into Muskegon Lake	1.4	1.6	12.3	2.5
49	White River	1.4	1.5	18.1	2.0
50	Pentwater River	5.0	2.7	5.8	12.5
52	Manistee River	<1	<1	5.9	2.5
53	Betsie River	1.3	1.2	6.2	
54	Platte River	2.5	1.4	9.9	5.0
60	Bear River, Petoskey	1.5	1.3	2.1	
62	Manistique River	11.1	1.6	6.5	
65	Escanaba River	<1	<1	16.1	2.0

NOTE: ng/l = ppt; 1 mile = 1.609 km.

TABLE 15

Lake Michigan River Water Pesticides--July 1969. (Adapted from The Lake Michigan Interstate Pesticide Committee of the Lake Michigan Enforcement Conference (103).)

Sample No.	Location	Total DDT	Dieldrin	BHC	Estimated PCB
		ng/l			
2	Big Suamico River	<10	<10	NC**	NC
3	Little Suamico River	<10		NC	NC
4	Pensaukee River	<10	<10	NC	NC
5	Oconto River	<10	<10	NC	NC
6	Peshtigo River	<10	<10	73.5	NC
7	Menominee River	<10	<10	128	NC
8	Mud Lake Creek	<10		16.0	NC
9	Clark Lake Creek	600*	10	17.8	NC
10	N Jacksonport Creek	<10		43.3	NC
11	Kangaroo Lake Creek	14.4*	<10	52.2	NC
12	Moonlight Bay	<10	<10	27.2	NC
13	Stony Creek	<10	<10	35.3	NC
14	Ahnapee River	360*	<10	72.2	NC
15	Kewaunee River	<10	<10	13.3	NC
16	East Twin River	<10	<10	50.0	NC
17	West Twin River	27.6	<10	14.4	
18	Manitowoc River	<10	<10	66.7	NC
19	Silver Creek	10.8	<10	34.4	NC
20	Calvin Creek	<10	<10	<10	NC
21	Pine Creek	574*	<10	<10	NC
22	Point Creek	<10	<10	<10	NC
23	Fisher Creek	<10	<10	<10	NC
24	Centerville Creek	<10	<10	174	NC
25	Seven Mile Creek	542*	<10	20.8	NC
26	Pigeon River	<10	<10	<10	NC
27	Sheboygan River	260*	<10	5.2	Very possible
28	Black River	97.4*	<10	18.4	Very possible
29	Sauk Creek	29.8*	<10	19.5	Possible
30	Milwaukee River	42.8*	28.8	44	Possible
31	Menomonee River	53.8*	14	84	Possible
33	Root Creek	10.0	<10	64	Possible
34	Pike River	49.1*	<10	10.4	Possible
35	Barnes Creek	75.5*	<10	<10	Possible
36	Calumet River at Calumet City	50*	<10	40.4	
37	Burns Ditch	<10	<10	17.9	NC
38	Trail Creek	93.6*	<10	47	Possible
39	Galien River	56.9*	<10	14	Possible
40	Drain at Sawyer	16.0	<10	40	
41	St. Joseph River	<10	<10	23.0	30.0
42	Paw Paw River	36.0*	45.2	7.2	
43	Black River	82.2*	<10	<10	NC
44	Kalamazoo River	<10	<10	33.0	NC
45	Black River	<10	<10	<10	<10
46	Pigeon River	<10	<10	10.0	NC
47	Grand River	11.4*	<10	<10	Possible
48	Muskegon River at mouth into Muskegon L.	<10	<10	<10	NC
49	White River	<10	<10	<10	NC
50	Pentwater River	<10	<10	<10	<10
51	Pere Marquette River	<10	<10	20	NC
52	Manistee River	24.4*	<10	<10	Possible
53	Betsie River	<10	<10	<10	NC
54	Platte River	<10	<10	<10	NC
56	Leelanau Lake	<10	<10	<10	NC
57	Boardman River	<10	<10	Inter	NC
59	Lake Charlevois Outlet	600*	<10	<10	NC
60	Bear River, Petoskey	<10	<10	NC	NC
61	Millecoquins Creek	<10	<10	NC	NC
62	Manistique River	<10	<10	<10	NC
63	Sturgeon River	<10	<10	<10	NC
64	Whitefish River	975*	<10	NC	NC
65	Escanaba River	<10	<10	<10	NC
66	Ford River	<10	<10	<10	NC

*Retention time not exact
**Not calculated
NOTE: ng/l = ppt

TABLE 16

Pesticides in Stream Sediments-July 1969. (Adapted from The Lake Michigan Interstate Pesticide Committee of the Lake Michigan Enforcement Conference, 1972 (103).)

Sample No.	Location	Total DDT	Dieldrin	Estimated PCB
		µg/g*		
1	East River	1.07	0.001	0.50
2	Big Suamico River	0.001	0.001	0.01
3	Little Suamico River	0.006	0.001	0.02
4	Pensaukee River	0.023	0.001	0.05
5	Oconto River	0.002	0.001	0.01
6	Peshtigo River	0.002	0.001	0.01
7	Menominee River	0.001	0.001	0.01
8	Mud Lake Creek	0.001	0.001	0.01
9	Clark Lake Creek	0.011		0.028
10	N Jacksonport Creek	0.003	0.001	0.01
11	Kangaroo Lake Creek	0.015	0.001	0.01
13	Stony Creek	0.019	0.001	0.03
14	Ahnapee River	0.102	0.001	1.10
15	Kewaunee River	0.033	0.001	0.035
16	East Twin River	0.079	0.001	0.21
17	West Twin River	0.035	0.001	0.052
18	Manitowoc River	0.037	0.001	0.12
19	Silver Creek	0.016		0.01
20	Calvin Creek	0.082	0.001	0.01
21	Pine Creek	0.008		0.033
22	Point Creek	0.026	0.001	0.015
23	Fisher Creek	0.042	0.001	0.021
25	Seven Mile Creek	0.087	0.001	
26	Pigeon River	0.053		0.036
27	Sheboygan River	0.173	0.001	7.2
28	Black River	0.011		0.052
29	Sauk Creek	0.067	0.001	0.065
30	Milwaukee River	0.082	0.040	3.2
31	Menomonee River	0.114	0.040	4.4
33	Root Creek	0.069	0.004	0.075
34	Pike River	0.137	0.002	0.20
35	Barnes Creek	0.010		0.013
36	Calumet River at Calumet City	0.063	0.004	1.25
37	Burns Ditch	0.017	0.001	0.021
38	Trail Creek	0.143	0.002	
39	Galien River	0.024	0.002	0.060
40	Drain at Sawyer	0.009		0.037
41	St. Joseph River	0.029	0.001	0.032
42	Paw Paw River	0.035	0.001	0.08
43	Black River	0.044	0.005	0.11
44	Kalamazoo River	0.053	0.001	0.043
46	Pigeon River	0.03	0.001	0.018
47	Grand River	0.061	0.002	0.17
48	Muskegon River at mouth into Muskegon Lake	0.006	0.001	0.01
49	White River	0.008	0.003	0.02
50	Pentwater River	0.001	0.001	0.01
51	Pere Marquette River	0.007	0.001	0.01
52	Manistee River	0.001	0.001	0.01
53	Betsie River	0.012	0.001	0.01
54	Platte River	0.011	0 001	0.01
56	Leelanau Lake	0.005	0.001	0.01
57	Boardman River	0.008	0.001	0.01
59	Lake Charlevoix	0.008	0.001	0.01
60	Bear River, Petoskey	0.006	0.001	0.01
61	Millecoquins Creek	0.002	0.001	0.01
62	Manistique River	0.114	0.001	0.80
63	Sturgeon River	0.003	0.001	0.01
64	Whitefish River	0.007		0.04
65	Escanaba River	0.069	0.001	1.46
66	Ford River	0.007	0.001	0.03

*Wet basis

NOTE: µg/g = ppm

low, most samples containing less than 10 ng/l (10 ppt). However, the stream sediments contained several hundred times more DDT and dieldrin than the associated tributary waters. Nonetheless, pesticide monitoring in lake sediments is often neglected although sediments serve as the most important repository of pesticides in aquatic ecosystems (105). Eroded soil particles containing adsorbed pesticides eventually settle on lake bottoms and become part of the sediment. In addition, pesticides in water or in aquatic organisms may return to the sediment through sorption and decomposition. These situations underscore the importance of including stream and near-shore sediment sampling in any monitoring program of pesticides. The pesticide pollution of Lake Michigan, as indicated by high accumulations of DDT and dieldrin in sediment, fish, and biological samples, suggests higher usage of pesticides in the watershed of this lake than of the other lakes. One of the suspected sources is from treated agricultural watersheds; however, definitive studies of the contribution of farming activities on the Great Lakes Basin are yet to be undertaken.

3.8.2 Projections and Assumptions

The pesticide contamination of Lake Michigan has already been well-established. Pesticide input to the Great Lakes will continue as long as pesticides are used in the watersheds. As indicated in Section 3.7, pesticide use is predicted to increase due to projected increased crop and livestock production per unit area and the absence of suitable alternatives to replace chemical pest control. Even with curtailment in the use of DDT and other organochlorine insecticides, residues of these chemicals still present a continuing hazard to the aquatic environment due to their long persistence in soils. Evidence indicates that there is already a "pool" of DDT in soils, especially in heavily treated areas such as orchards. It is estimated that, if the use of DDT was to cease in the mid-1970's, detectable levels will still be found in the troposphere and the oceanic mixed layer in the year 2000 (207). Build-up of residues from herbicides may result due to increasing use and repeated applications. Commonly used herbicides like the s-triazines are known to persist in soils for a year or more. The less

persistent organophosphorous and carbamate insecticides may pose less of a hazard from residues. However, because of their rapid degradability in soils, more frequent applications are needed to obtain effective insect control. Metabolite accumulation in soil is relatively unknown except for DDE and TDE (DDD).

The seriousness of future pesticide pollution problems in the Great Lakes depends heavily on the adequacy of erosion and sediment control measures. Surface runoff--implicated as the main avenue of pesticide entry to aquatic systems--can be controlled effectively by present technology. However, putting the technology to work is often hampered by inadequate control measures and lack of effective implementation.

In the Great Lakes Region about 7,756,300 ha (19,165,900 acres) or 60% of the total area is currently in need of land treatment (Table 17). Using current conservation practices embodied in Public Law (P.L.) 566 and P.L. 46, only 40% is projected to be treated by the year 2020, leaving 60% untreated. Even accelerated land treatment schemes which involve cost-sharing between the Federal government and the farmer are projected to treat only 44% of the land by the year 2020.

It is interesting to note that in the above control legislation, the erosion and sediment problem has to be solved primarily on a voluntary basis without governmental units providing adequate regulations. These control programs are extremely slow and, if they continue to be implemented in this voluntary fashion, abatement of pesticide pollution of aquatic ecosystems will be very limited in the next 50 years. Hazards of erosion will be intensified by the opening of idle cropland since most of these lands lie on steep slopes or have shallow soils.

In order to minimize pesticide pollution effectively, mandatory sediment control programs in the Great Lakes Region are required urgently.

3.9 Review of Institutional Arrangements to Regulate Pesticide Pollution

The Federal Water Pollution Control Act Amendments of 1972 broadly regulate water pollution from all sources, including such non-point sources

TABLE 17

Total area, area treated and area needing treatment for 1970 and projected area that will be treated under current agricultural conservation practices and recommended accelerated programs in the U.S. Great Lakes Region, for 1980, 2000 and 2020. (Excerpted and recalculated from Appendix 13, Land Use and Management Work Group of the Great Lakes Basin Commission, sponsored by the Forest Service and Soil Conservation Service of the U.S. Department of Agriculture (66).)

Type of agricultural land	Total area 1970		Area treated prior to 1970		Area needing treatment after 1970	
	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3
Cropland	11,577.9	28,609.0	4,800.8	11,862.7	6,777.1	16,746.3
Pasture	1,418.8	3,505.8	439.6	1,086.2	979.2	2,419.6
TOTAL	12,996.7	32,114.8	5,240.4	12,948.9	7,756.3	19,165.9

	Projected area that will be treated under current conservation laws*						Area needing treatment after 2020	
	1980		2000		2020			
	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3
Cropland	650.1	1,606.4	1,950.3	4,819.3	2,730.7	6,747.8	4,046.3	9,998.5
Pasture	94.9	234.6	284.9	704.1	398.9	985.9	580.2	1,433.7
TOTAL	745.0	1,841.0	2,235.2	5,523.4	3,129.6	7,733.7	4,626.5	11,432.2
% **	9.6		28.8		40.4		59.6	

	Projected area that will be treated if the recommended accelerated programs are approved +						Area needing treatment after 2020	
	1980		2000		2020			
	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3	Ha x 10^3	A x 10^3
Cropland	779.9	1,927.2	2,145.0	5,300.4	2,990.6	7,389.9	3,786.5	9,356.4
Pasture	114.0	281.7	313.5	774.6	437.9	1,082.0	541.3	1,337.6
TOTAL	893.9	2,208.9	2,458.5	6,075.0	3,428.5	8,471.9	4,327.8	10,694.0
% **	11.5		31.7		44.2		55.8	

*Public Law 566 and P.L.-46 programs

**1970 used as base

+Programs providing funds for installation costs of land treatment measures

as agricultural land. In terms of pesticide pollution, specific federal and state laws and regulations have been enacted to control the registration, distribution, use and application of pesticides.

The Federal Environmental Pesticide Control Act (FEPCA) was enacted on October 21, 1972, amending the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) of 1947. Among the major provisions are included: registration, classification, applicator certification, intensified research and monitoring programs, development of disposal policies, and strengthening of enforcement policies. Most of the provisions of the act became effective immediately, while others have deadlines for later enforcement pending the establishment of regulations and development of federal standards to guide states in implementing the legislation. All provisions must be implemented by October 1976. Among the provisions to take effect by 1976 are: 1. classification of pesticides into general and restricted use, and 2. development, by states, of certification and licensing programs for applicators of restricted pesticides and subsequent approval of such programs by the U.S. Environmental Protection Agency (U.S. EPA). The act emphasizes the immediate need to develop biologically-integrated alternatives for pest control.

By authority of the FIFRA, the U.S. EPA Administrator after extensive hearings cancelled the nation-wide sale and use of DDT in June 1972 except for health reasons; the order became effective December 31, 1972. Use of other hazardous pesticides, such as 2,4,5-T, aldrin and dieldrin, have been cancelled temporarily, pending the results of public hearings.

The Great Lakes States have current laws or regulations which either conform with or are more stringent than the federal act. Given below are the laws and regulations by state. Most of the information given is based on that compiled by the Great Lakes Water Quality Board (67) and U.S. EPA (48).

3.9.1 Illinois

The Pesticide Control Law (Ch. 5, Ill. Ann. Stat. 1969) was passed

on June 25, 1969. This act provides for labeling of pesticides to prevent contamination of waters and the environment by regulating, restricting or prohibiting the sale, use or application of pesticides. It authorizes the Illinois Department of Agriculture to register and label all pesticides sold in the state. The agencies responsible for rules and regulatory programs are the Departments of Agriculture and Public Health.

Effective January 1, 1970, the Director of the Department of Public Health prohibited the sale, use or application of DDT without permit.

Illinois was among the first states to require the licensing of custom applicators of pesticides as provided in the Customs Application of Pesticides Act (Ch. 5, Ill. Ann. Stat. 1966).

3.9.2 Indiana

In 1971, Indiana enacted a law regulating the distribution, sale and use of pesticides and providing for the creation of a Pesticide Review Board (Sec. 15-2701, Ind. Ann. Stat. Supp. 1972). The law provides that all pesticides sold and transported in the state be registered with the Office of the State Chemist.

The Pesticide Review Board, which is composed of members from state agencies as well as from public and industry sectors, after public hearings, can regulate and adopt a list of restricted-use pesticides and pesticides for use by prescription only. Regulatory functions rest with the State Chemist and Pesticide Review Board.

The Indiana Stream Pollution Control Board works closely with the State Chemist and Pesticide Review Board to insure proper pesticide application in order to control pollution of streams in the Great Lakes Basin.

3.9.3 Michigan

Michigan promulgated the Economic Poison Law in 1967 (Sec. 12.352, Mich. Ann. Stat. 1967). This law specifies the registration and labeling of all economic poisons sold in the state. It further regulates the transportation, distribution, sale, and labeling of economic poisons. In the

same year, the Pesticide Applications Law (Sec. 12.353, Mich. Ann. Stat. 1967) was passed making it unlawful to apply pesticides commercially unless licensed by the Michigan Department of Agriculture.

An Economic Poisons Advisory Committee was formed in 1970. Under the interagency Agreement for Economic Poisons Label Review, each state agency makes periodic reviews of all labels of economic poisons proposed for registration.

In March 1972, a restricted-use pesticides regulation was promulgated for the identification of restricted-use pesticides. The restricted pesticides are to be sold only by licensed dealers to licensed applicators or representatives of governmental agencies. The Michigan Department of Agriculture assumes the regulatory responsibility for proper pesticide usage.

3.9.4 Minnesota

In 1969, the Economic Poisons and Devices Law (Sec. 24.069, Minn. Stat. 1969) was promulgated, creating the Pesticide Advisory Board which developed regulations on restricted-use pesticides issued by the Minnesota Department of Agriculture in mid-1970. These regulations restricted the sale, use and distribution of DDT and several other compounds. The agency responsible for registration and labeling is the Department of Agriculture while the Departments of Agriculture and Natural Resources assume the regulatory functions. Licensing of commercial applicators and dealers of restricted-use pesticides by the Department of Agriculture is provided in the Spraying and Dusting Law (Sec. 18.031, Minn. Stat. 1969).

3.9.5 New York

Enactment of the Economic Poisons Law in 1973 (Sec. 33-0101, N.Y. Environ. Con. Law, 1973) required that all pesticides used in the state be registered with the New York Department of Environmental Conservation. The department also has restricted the use of certain pesticides and prohibited the use of others. The Custom Application of Pesticides Act of 1973

(Sec. 33-0901, N.Y. Environ. Con. Law 1973) makes it unlawful to engage in custom application and sale of restricted-use pesticides unless registered and licensed by the Department of Environmental Conservation.

3.9.6 Ohio

Regulatory control of pesticide usage and application procedures is maintained by the Ohio Department of Agriculture. Pesticides must be registered with the department as provided by the state's Economic Poison Act (Sec. 921.11, Ohio Rev. Code, 1954) and Herbicide Law (Sec. 921.06, Ohio Rev. Code Supp. 1954). In addition, the Ohio Pesticide Use and Applicator Law (Sec. 921.41, Ohio Rev. Code Supp. 1971) requires the licensing of pesticide applicator firms as well as certification of pesticide equipment operators by the Department of Agriculture.

3.9.7 Pennsylvania

Registration of pesticides used in the state is embodied in the Pesticide Act of 1957.

3.9.8 Wisconsin

The basic Wisconsin pesticide use and control legislation was embodied in Chapter 94 of the Statutes. By authority of this law the Wisconsin Department of Agriculture has promulgated rules under Chapter Ag 29 of the Administrative Code to regulate registration and labeling of pesticides.

The Wisconsin Statutes of 1969 contained two laws on pesticides in Chapters 146 and 426. Chapter 426 prohibits the sale, use and transportation of DDT except under emergency conditions, making Wisconsin the first state to ban DDT by statute. Chapter 146 created the Pesticide Review Board, bringing three state departments--Agriculture, Natural Resources, and Health and Social Services--in close coordination in developing regulatory programs concerning pesticide use restrictions under the advisement of a Technical Advisory Council. The Administrative Code Chapters Ag 29 and NR 80 were

products of such interagency cooperation. Ag 29 lists certain prohibited-use pesticides and six restricted-use pesticides for agricultural and other purposes. NR80, promulgated by the Department of Natural Resources, lists restricted-use pesticides that may be used by permit only for wild animals.

Enforcement of statutes and rules is performed cooperatively between the Departments of Agriculture and Natural Resources although the latter has general jurisdiction.

3.9.9 Great Lakes Basin

The Great Lakes States appear to have adequate existing laws and regulations to meet the standards of the Federal Act concerning the sale, registration, distribution, use, and application of pesticides. Except for Wisconsin and Pennsylvania, most of the states already have passed the necessary enabling legislation requiring the licensing of commercial applicators of restricted-use pesticides. A proposed bill of this nature is pending in the Wisconsin Legislature.

The primary limitation in the success of pesticide regulations is in the monitoring and enforcement areas. Progress in any monitoring and enforcement program is curtailed severely by lack of funds and personnel. In addition, rapport between agencies responsible for enforcement programs is not always possible; thus, one single agency should be given sole authority for implementing all pesticide regulations.

One critical weakness of pesticide regulatory programs has been the inability to estimate quantities of pesticides used and location of their use. In order to have a sound pesticide management and control program, the state agencies concerned must establish a system of documenting the amounts and levels of pesticides used as well as information relating to when and where they were used. Furthermore, state agencies should initiate a vigorous effort to include soil and sediment in any monitoring program undertaken.

Successful curtailment of pesticide pollution from agricultural land can be achieved only if mandatory erosion- and sediment-control programs are enacted by all levels of state governments.

Section 4

SUMMARY OF RELEVANT ONGOING OR PROPOSED RESEARCH, DEMONSTRATION OR MONITORING PROGRAMS

4.1 Ongoing Research

There is a wealth of ongoing research covering many aspects of pesticides in relation to water quality. Basic research is pursued vigorously to gain better insight into the movement, degradation, and metabolic pathways of pesticides in soils as well as the fate and behavior of pesticides in aquatic systems. Analytical procedures are constantly undergoing modifications and improvements in order to better identify and quantify the pesticide residues and their metabolites in the environment. No attempt is made to list all the projects in progress, but a cross-section of the diverse research activities with particular reference to the Great Lakes Basin is presented in Table 18.

The assessment and quantification of the contribution of agricultural watersheds by runoff to the pesticide pollution of the Great Lakes are important aspects of the projects in progress (Table 18). In some projects, movement of pesticides from soils to aquatic systems will be evaluated by systems analyses in an attempt to develop a mathematical model for pesticide transport and runoff from agricultural watersheds. Monitoring and surveillance programs will continue to provide pesticide levels and rates of change of pesticide levels in tributary and lake waters.

The USDA is continuing support of projects related to alternative methods of pest control with the ultimate objective of minimizing residues in the environment. Alternative methods being pursued include biological control, plant resistance, and integrated approaches. Additionally, better pesticide management techniques, such as judicious methods of application and use of safer pesticides, are being investigated.

4.2 Proposed Projects

Two proposed projects which may have significant impact on Great Lakes water quality management follow (Table 19). These projects, including the

TABLE 18

Recent and current research on pesticides in the United States with particular reference to the Great Lakes Basin

Program Title	Investigator/Affiliation	Duration	Sponsor
Chemical and sediment movement from agricultural land into Lake Erie	G. O. Schwab Ohio State University		USDI
Evaluation of pesticide sources and levels tributary to Lakes Michigan and Superior	L. A. Lueschow Wisconsin Dept. of Natural Resources		U.S.EPA
Pesticide inputs and levels in Minnesota waters of the Lake Superior Basin	T. A. Olson University of Minnesota		U.S.EPA
Pesticide movement from cropland into Lake Erie	A. C. Waldron Ohio State University	1971-72	U.S.EPA
Pesticide runoff in the Great Lakes Basin	B. G. Ellis Michigan State University	1972-73	U.S.EPA
Quantification of pollutants in agricultural runoff	J. N. Dornbush South Dakota State University	1972-73	U.S.EPA
Development of pesticide transport and runoff model	N. Crawford Hydrocomp, Inc.	1972-73	U.S.EPA
Attenuation and runoff of pesticides from agricultural lands to surface waters	J. Box USDA-SWCRD Watkinsville, Georgia	1972-73	U.S.EPA
The study and use of soil parameters for describing pesticide movement through soils	J. M. Davidson Oklahoma State University	1972-74	U.S.EPA
Pesticide monitoring program - Lake Michigan and tributaries - Illinois	B. J. Leland Illinois Dept. of Public Instruction		U.S.EPA
Great Lakes pesticide monitoring program, Indiana	R. L. Carter Indiana State Board of Health		U.S.EPA
Monitoring of pesticide levels in the Great Lakes	R. Reinert U.S. Dept. of Commerce Lansing, Michigan		U.S. Dept. of Commerce
Pesticide monitoring of aquatic environment, Michigan portion of the Great Lakes Basin	C. H. Fetterolf Michigan Dept. of Natural Resources		U.S.EPA
Environmental implication of pesticide usage	Five universities cooperating on a regional project	1965-75	USDA-CSRS
Herbicide movement from application sites and effects on non-target species	Nine universities and agencies cooperating on a regional project	1970-76	USDA-CSRS
Pesticide mobility and degradation in soil-water systems	Nine universities cooperating on regional project	1965-74	USDA-CSRS
Analytical methods for pesticides in soil and water	F. A. Gunther University of California-Riverside		State of California
Development of analytical chemical methods for environmental and water quality	H. A. Salman U.S. Dept. of the Interior Denver, Colorado		USDI
Economic and social impact of adjustment in use of pesticides	R. B. Norgaard University of California-Berkeley	1971-76	USDA-CSRS
Weed control practices to reduce pollution	M. A. Ross Purdue University	1970-75	USDA-CSRS
Chlorinated pesticides in the soil-water plant systems and their management to avoid pollution	F. G. Viets, Jr. USDA-SWCRD Fort Collins, Colorado	1968-73	USDA-CSRS
Pesticide detoxication mechanisms of microorganisms in soil and water	J. M. Bollag Pennsylvania State University		USDA-CSRS
Ecology of pesticides in an aquatic ecosystem	R. C. Ball Michigan State University	1964-69	USDA-CSRS
Distribution of organophosphorus pesticide residues in natural waters and sediments	M. L. Yates U.S. Dept. of the Interior		USDI
Black Creek study, Maumee River Basin, Allen County, Indiana	Allen County, Soil and Water Conservation District	1973-77	U.S.EPA

TABLE 19

Proposed projects with reference to Great Lakes pollution

Program Title	Investigator/Affiliation	Duration	Sponsor
Watersheds Study - Task C International Joint Commission, Land Use Activities Reference Group	Several institutions and agencies of the states of Indiana, Michigan, New York, Ohio and Wisconsin and of the Canadian Federal Government and Ontario Provincial Government	1974-78	International Joint Commission through U.S. EPA and the Governments of Canada and Ontario
Development and Implementation of a Sediment Control Ordinance: Institutional Arrangements Necessary for Implementation of Control Methodology on Urban and Rural Lands	T. C. Daniel University of Wisconsin-Extension	1974-78	U.S. EPA

ongoing research in Allen County, Indiana, recognize the paramount importance of a multidisciplinary and multiagency approach to achieving solutions to water pollution problems arising from non-point sources. The major objectives of the IJC-Land Use Activities Reference Group watersheds study are to: 1. investigate the effects of land drainage on the pollutional input to the Great Lakes, 2. develop a predictive capacity with respect to the sources, forms and amounts of pollutants reaching the Great Lakes Basin, and 3. develop remedial measures for maintaining and improving water quality in the Great Lakes. The overall objectives of the project coordinated by the Wisconsin Board of Soil and Water Conservation Districts and the University of Wisconsin are to: 1. demonstrate the effectiveness of land control measures in improving water quality, and 2. devise the necessary institutional arrangements for the preparation, acceptance, adoption and implementation of a sediment control ordinance applicable to incorporated and unincorporated areas on a county-wide basis. Outputs from these projects, in addition to answering the objectives, will aid the participating U.S. federal and state and Canadian federal and provincial governments to implement portions of their water quality laws.

Section 5

NATURE AND AVAILABILITY OF TECHNOLOGY TO COPE WITH POLLUTION PROBLEMS

Technological competence is available to minimize pesticide contamination of soil and water, reduce pesticide transport to aquatic environments, and improve detection of pesticide pollution. However, success of a particular technology may be limited by the lack of mandatory regulations to implement it, such as is the case with erosion and runoff control programs.

It is important to minimize soil pesticide residues since the magnitude of pesticide pollution of aquatic ecosystems is related directly to the amounts present in soils. This is best done by reducing dosage through proper application methods. Drift loss does not only increase the amount of compound applied but also contaminates nontarget areas adjacent to application sites. Aerial spraying during favorable weather conditions should become a rule of thumb for applicators. Other means of reducing build-up of residues in the soil involve the employment of alternative methods of pest control and less persistent compounds. Practical alternative methods, however, are yet to be developed.

Most residues remain in the plow-layer depth either adsorbed or absorbed by particulate matter and are susceptible to transportation during occurrence of runoff. Accelerated erosion of the pesticide-soil complex can be minimized by judicious use of soil management and conservation practices. Federal and state agencies are involved in controlling erosion, but programs have been carried out on a voluntary incentive basis. While soil and water losses can be estimated using the universal soil loss equation, mathematical models for pesticide transport from agricultural watersheds need development. Such models would entail understanding the mechanisms of the pesticide-soil-biota-water interactions in the watersheds.

Monitoring of pesticide pollution in the Great Lakes is conducted by the U.S. federal agencies in cooperation with the lake states. In addition, the International Joint Commission and Canadian government conduct extensive

water quality assessment. Monitoring activities center mostly on the aquatic environment, but little or no research activity is being centered on residue detection in agricultural soils and in the atmosphere overlying the Great Lakes. Apparently, development of a technique to measure the amount of pesticide in the atmosphere should be one important aspect of the monitoring programs being undertaken.

Section 6

NEED FOR NEW RESEARCH, DEMONSTRATION OR MONITORING PROGRAMS

Pesticide pollution of the Great Lakes is well-established. Runoff from agricultural lands in the watersheds has been implicated as a major source of residues and/or metabolites. This suspicion is based largely on the fact that the major portion of pesticide usage occurs in crop production. It becomes imperative, therefore, to assess the contribution of agricultural lands to the total pesticide load of the Great Lakes and associated tributary rivers. Only very recently have there been attempts made to identify clearly the sources of pollutant input (including pesticides) to the Great Lakes through integrated multidisciplinary and multiagency approaches using large watersheds, as indicated in Section 4. These research and/or demonstration programs will take at least 5 to 10 years to be realized or initiated depending upon funding potential.

Specific research needs are as follows but not arranged necessarily in any order of priority.

a. Pesticide inputs on agricultural lands. This involves a systematic documentation of the lands and amounts of pesticides used, category of crops treated, and area treated for each crop category.

b. Pesticide degradation rates and fate of toxic metabolites. Little information is available on the fate of the metabolites of persistent pesticides in soil and aquatic systems. Likewise, information about the degradation of newer compounds, which are finding increasing use due to the restrictions placed on the persistent pesticides, is needed for the environmental conditions found in the Great Lakes Basin. Furthermore, the fate and toxicity of their metabolites need elucidation.

c. Transport of pesticides from agricultural watersheds to the Great Lakes. More information is needed on the mechanisms of pesticide movement from varied types of soils and cropping pattterns present in the watersheds. Inputs of pesticides to the Great Lakes by atmospheric fallout or rainout are often neglected.

d. Expansion and redesigning of monitoring programs. Sediments act as reservoirs of pollutants (including pesticides), but they are often overlooked in monitoring programs. Inclusion of soil, sediment, and atmospheric samplings in the monitoring programs is highly desirable. Critical evaluation of sampling, sample preservation and analytical techniques should be made as often as possible.

e. Search for improved pest control techniques. Since pesticide usage in crop production is inevitable in the foreseeable future, considerable research effort should be directed towards the development of pesticide management techniques, including application methods and development of safer pesticides compatible with efficient soil and crop management systems. However, continuing and new research techniques of pest control are needed to provide alternatives to chemical pest control.

Section 7

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Category A6 - Nutrients

POLLUTION OF THE GREAT LAKES BY NUTRIENTS FROM AGRICULTURAL LAND

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To be used as portion of the U. S. Task A Report on GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by the International Joint Commission - prepared in partial fulfillment of U. S. Environmental Protection Agency Contract No. 68-01-1598

July 1974

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Section 1

SUMMARY

1.1 Introduction

The land use category on Nutrients - Agricultural Land is concerned with the nutrient contribution to the Great Lakes from agricultural land. The nutrients of importance are phosphorus and nitrogen, as algal growth is considered to be limited by one or the other of these nutrients in most lakes. Phosphorus reduction has been emphasized in municipal waste treatment. Consequently, assessment of the phosphorus loading from agricultural land is of particular importance. Relatedly, a need exists to determine the availability of the information required to assess the problem.

1.2 Summary of Findings and Conclusions

The information on nutrient transport from agricultural lands was evaluated to obtain estimates of nutrient input rates to the Great Lakes per unit of land according to crop use category. Data based on runoff from small field plots gave usable information on soluble nutrient transport but overestimated total nutrient transport as a large portion of the eroded soil would not reach the Great Lakes. However, this data allowed comparison of various crop use categories. The most reliable data for estimating nutrient transport to lakes was based on streams draining primarily agricultural areas. However, this data varied considerably among investigations and did not allow evaluation of crop use categories. Based on streams draining agricultural lands, the phosphorus loading was estimated to be about 0.4 kg per hectare (0.35 lb per acre) for total P and 5 kg per hectare (4.5 lb per acre) for total N. For phosphorus, the estimated contribution from agricultural land was about 20 percent of the total phosphorus loading to the Great Lakes. Total nitrogen loadings for the Great Lakes were not available. Because of the range in the data used, the above estimates may be inaccurate.

Within the range of available data, differences in transport rates for soluble nutrients were not found for the different crop use categories. Total nutrient transport from field plots was related to crop cover and corresponding soil erosion. Estimated nutrient loadings were related more to land area than differences in input rates for different crop categories.

The area devoted to agricultural land in the Great Lakes Basin will not increase appreciably in the future. Consequently, any increase in nutrient transport to the Great Lakes from agricultural land will apparently result only if more intensive agriculture increases nutrient losses. In this regard, the effect of fertilization is important. At present, quantitative analysis of the effect of fertilizer use on a basin-wide basis can not be made.

Research and demonstration projects are needed to more accurately assess the problem and establish adequate control measures. Monitoring of representative agricultural drainage basins is needed to provide quantitative information on nutrient transport as related to land use and other important factors. Research is needed on the factors and mechanisms controlling nutrient transport from agricultural lands, especially for soluble nutrients, to determine the role of soil properties, fertilizer use and other factors in controlling nutrient concentrations in runoff. Transport in streams and rivers should be evaluated to provide information on losses due to interaction with stream sediments. The effects of particulate nutrients on the nutrient status of lake surface waters should be determined to clarify the importance of including particulate nutrient forms in lake nutrient budgets. Demonstration projects are also needed to establish the effectiveness of control measures, such as soil conservation and erosion control programs, and the feasibility of implementing control policies and programs.

Section 2

SUPPORTING MATERIAL

2.1 Introduction

Eutrophication is regarded as one of the most important water quality problems for the Great Lakes. Phosphorus and/or nitrogen are considered to be the nutrients limiting the growth of algae in most lake waters, and control of these nutrients has been emphasized as the approach for controlling eutrophication. In particular, phosphorus reduction has been selected as the method of control because reduction of the phosphorus loading is considered more feasible than nitrogen control. Progress has been made in reducing the phosphorus loading through treatment of municipal waters.

The large area of agricultural land in the Great Lakes Basin raises concern over the nutrient load contributed by agricultural lands as well as other diffuse sources. Even a relatively small input per unit land area could result in a relatively large nutrient input. Consequently, a need exists to evaluate the impact of agricultural land on the nutrient loading of the Great Lakes. Among the goals of this review were to evaluate the nutrient loading to the Great Lakes from agricultural lands, and to determine the availability of information needed to conduct this evaluation.

2.2 Scope of Study

An attempt was made to gather substantially the relevant information available relating to nutrient transport from agricultural lands. Information was obtained from scientific publications, project reports, and state and federal reports (see Section 7). The Great Lakes Basin Framework Study was an important source of information on the Basin. The report of the Great Lakes Water Quality Board (1973) provided information on the Great Lakes, particularly estimates of total phosphorus loadings.

Information available to the present time was utilized, including information on ongoing research when it could be obtained.

2.3 Study Procedure

Information on nutrient transport from agricultural land was gathered and evaluated with the ultimate goal of providing information on the contribution of agricultural lands to the nutrient loading of the Great Lakes. An intensive survey of the scientific literature was conducted. Reports from research and demonstration projects were utilized. Letters were sent to individuals and organizations throughout the country thought to have relevant information. These letters requested publications and other information on the subject under investigation. University and state experts were consulted. Information on research in progress was obtained from Smithsonian Science Information Exchange.

The various assumptions made are discussed in the appropriate sections of the review. The assumption of general importance made was that the only nutrients of importance regarding effects on the Great Lakes were phosphorus and nitrogen.

Section 3

REVIEW OF FINDINGS OF PREVIOUS STUDIES

3.1 General Description Of Land Use Category

The category on Nutrients - Agricultural Land is concerned with agricultural land use - water quality relationships. The primary focus is on nutrient loadings to the Great Lakes from agricultural lands as related to type of agricultural land use. The categories of agricultural land use considered are a) row crops, b) close-grown crops, c) pasture and meadows, d) orchards and vineyards, and e) idle cropland.

3.2 Length of Time Activities in Practice

Land settlement of the Great Lakes Area began about 1820. Except for small cultivated areas, the region was nearly all forested. A major change in land use came with land settlement, land clearing, and shifting of much of the land into farms (EPA, 1971). By 1850, 1.8 million hectares (4.4 million acres) were in farms with 0.8 million hectares (1.9 million acres) in improved farm land. The area of improved farm land had increased to 5.3 million hectares (13 million acres) by 1920. In 1966, the total cropping area, including idle cropland, was about 11.6 million hectares (28.6 million acres). Land clearing and farming resulted in increased runoff, soil erosion, and nutrient loading to surface waters from land within the region. Although modern farming and soil conservation practices reduce the runoff and erosion problem, concern exists over whether current agricultural practices are adequate to protect water quality in the Great Lakes.

3.3 Types and Nature of Nutrients Associated With Agricultural Runoff

The category on Nutrients - Agricultural Land is concerned with nutrients considered to be of importance to the water quality of the

Great Lakes. Phosphorus and/or nitrogen have been implicated as the nutrients limiting algal growth in various lakes (Vollenweider, 1968). In the Great Lakes, the major focus has been on phosphorus, due in part to the greater possibility of phosphorus control. Various other nutrients essential to plant growth are contained in agricultural runoff, but there is little evidence that nutrients other than nitrogen and phosphorus limit algal growth in lake waters.

Phosphorus and nitrogen occur in various chemical and physical forms which affect their mobility in agricultural runoff. These relationships are discussed in sections 3.5 and 3.6 of this review.

3.4 State-of-the-Art In Assessing and Quantifying Nutrient Loadings to the Great Lakes from Agricultural Lands

3.4.1 Nutrient Contributions from Agricultural Lands[1]

The transport of nutrients from agricultural lands to the Great Lakes could occur along innumerable pathways and could involve many transport mechanisms. For purposes of this report, water is considered to be the primary transport vector, although it is recognized that nutrient loss by wind-blown particulate matter could be large in some instances.

Three data grouping were prepared which describe the flux of nitrogen and phosphorus from agricultural lands 1) seepage studies, 2) runoff studies and 3) drainage area studies. The first two study types refer to the transport of nutrients across the boundaries of land areas. The latter type refers to nutrient transport by continually flowing streams. It is important to recognize that only a few of the investigations cited were conducted to quantify nutrient runoff from watersheds to lakes. The objective of most studies was to measure nutrient loss from distinct land parcels. Therefore, questions of subsequent nutrient transport must be addressed before the data can be used for estimating nutrient contributions

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[1] This section is based on the report by Uttormark, Chapin and Green (1974).

from agricultural lands.

Seepage studies include lysimeter work and analyses of tile-drained fields. Data from these investigations may be useful in estimating the transport of nutrients from surface soils to groundwater aquifers, but the applicability of these results to the estimation of lake loadings depends greatly on the extent of groundwater-surface water exchange in the basin under consideration.

Losses of nutrients by storm runoff have been quantified in a number of studies. However, the ability to predict nutrient loadings from these data is again limited by the difficulty in defining the probability to transport from agricultural lands to lakes. This difficulty is amplified by the fact that a large portion of the nutrients lost from agricultural lands are associated with particulate matter which may settle out at intermediate points along the flow path, especially during overland flow or in intermittent stream channels.

From the standpoint of lake loadings, some of the most useful data are provided by drainage area studies because of the clearly defined pathway for nutrients contained in continually-flowing streams. However, some trade-offs are involved. Land use patterns are defined clearly for most seepage and runoff studies, but subsequent transport of nutrients to a lake is questionable. In contrast, nutrient flow paths are more clearly defined for drainage area studies, but descriptions of land use are very imprecise (i.e., "typical rural environment" or "mixed farmlands and woodlots").

3.4.1.1 Nutrient Losses by Seepage

Nutrient loss from agricultural lands by seepage through the soil profile has been a topic of interest for several decades. The majority of the earlier studies were conducted to evaluate alternative agricultural practices, such as crop rotation, fertilizer usage, plowing techniques, etc., and losses of nitrate nitrogen were studied almost exclusively. In more recent years, emphasis has been placed on evaluating the effects of agricultural practices on water quality, and losses of phosphorus as well as nitrogen have been reported.

Two types of studies are reported in the literature which describe the transport of nutrients by seepage through soils. These involve the use of lysimeters or the analysis of waters which flow from tile-drained croplands.

Lysimeters are constructed by surrounding a volume of earth on the sides and bottom with an impermeable material. The enclosed soil is more or less disturbed, depending on how it is placed in the enclosure. The top is exposed to the elements, and drains are connected to the bottom, so that all water percolating through the soil profile can be collected and analyzed. Runoff and erosion are generally prevented by the design of the unit. Surface areas of these units range from 0.1 to 10m^2 (0.009 to 0.9 ft^2), and it is reported that some units have been maintained actively for periods as long as 35 years.

In some parts of the country, particularly those regions in which irrigation is practiced, large tracts of land are underlain with tiles or other subsurface collection systems. Some of the irrigation water percolates through the soil profile and is removed by these drainage systems. A number of studies have been reported in which these drainage waters were sampled periodically and analyzed for nitrogen and phosphorus to measure nutrient losses. In many respects, studies of this type are similar to lysimeter studies, but the surface area covered are much larger, and not all of the seepage water is collected--some is lost to deeper aquifers.

The data from seepage studies presented in this section are grouped according to crop, but crop cover plays only a partial (and possibly minor) role in influencing nutrient loss.

Bolton, Aylesworth, and Hore (1970) measured nutrient losses in tile drainage effluent from twelve 0.1 ha (0.25 ac) plots at Woodslee, Ontario (clay soils). Seepage flows were recorded continuously, and effluent samples were filtered to remove sediment and analyzed for total nitrogen and phosphorus. The authors concluded that the nutrient loss was influenced predominantly by the amount of water that percolated through the soil.

Crop	Nutrient loss (kg/ha/yr)* N No fert.	N Fert.	P No fert.	P Fert.
Rotation:				
Corn	5.6	15.1	0.13	0.24
Oats and alfalfa	4.3	5.7	0.13	0.13
Alfalfa, 1st year	4.8	3.9	0.13	0.15
Alfalfa, 2nd year	4.7	8.6	0.08	0.22
Continuous:				
Corn	6.6	14.0	0.26	0.29
Blue grass sod	0.3	0.7	0.01	0.12
Mean:	4.4	8.1	0.12	0.19

*kg/ha/yr x 0.89 = lb/ac/yr

In contrast, Sylvester and Seabloom (1962) monitored irrigation return flows in the Yakima Valley in Washington and found that more nitrate and soluble P were lost during the 6-month non-irrigation period than during the irrigation season even though 130 cm (51 inches) of water were applied (average rainfall was 18 cm (7 inches)/yr). The return flow consisted of both surface runoff and seepage through subsurface drains which continued to flow during the non-irrigation season. Samples were collected and analyzed for nitrate, total Kjeldahl nitrogen, and dissolved and total phosphorus. This project was also

	Nutrient loss (kg/ha)* Irrigation season	Non-irrigation season	Annual
Nitrate - N	34.	39.	73.
Dissolved - P	0.78	1.7	2.5

*kg/ha x 0.89 = lb/ac

described by Sylvester (1961) and separate results are given for surface and subsurface drains.

	Nutrient loss (kg/ha/yr)*			
	Total - N		Total - P	
	Range	Mean	Range	Mean
Surface drains	2.8-27	(16)	1.0-4.4	(2.5)
Subsurface drains	61-186	(103)	3.8-10	(7.7)

*kg/ha/yr x 0.89 = lb/ac/yr

Studies in which nutrient losses from croplands by tile drainage were compared to losses via surface runoff were also conducted in California and Idaho.

Johnston, Ittihadich, Daum and Pillsbury (1965) analyzed tile drainage effluent and surface runoff from irrigated land in the San Joaquin Valley of California. Four 19 to 60 ha (47 to 148 ac) plots growing cotton, alfalfa and rice were studied. The soils were heavy silty clays, and tile depth averaged 190 cm (75 inches). Analyses were made for total nitrogen and total phosphorus. No estimates were made as to the quantity of nitrogen or phosphorus lost through deep percolation.

	Nutrient loss (kg/ha/yr)*		Fertilizer applied (kg/ha/yr)*	
Crop	Total - N	Total - P	N	P
Cotton and rice			300	52
Tile drainage	110.	0.19		
Surface runoff	11.	0.81		
Cotton			220	36
Tile drainage	13.	0.05		
Surface runoff	6.4	0.17		
Alfalfa			none	none
Tile drainage	3.5	0.08		
Surface runoff	1.6	0.20		
Rice			94	none
Tile drainage	42.	0.60		
Surface runoff	5.2	0.11		

*kg/ha/yr x 0.89 = lb/ac/yr

Carter, Bondurant, and Robbins (1971) measured nitrate and dissolved phosphorus in surface runoff and subsurface (tile) drainage from an 82,000 ha (202,600 ac) tract in southern Idaho. The major crops were alfalfa, beans, grain, sugarbeets, corn, and pasture. Precipitation averaged 21 cm (8.3 inches) per year and irrigation supplied an additional 200 cm (79 inches) of water annually.

Nutrient inputs in kg/ha/yr*	N	P
Fertilizer	60.	30.
Irrigation water	2.3	1.0
Precipitation	negligible	negligible
Nutrient losses in kg/ha/yr*	**NO_3-N**	**Sol-P**
Surface runoff	0.35	0.17
Tile drainage	35.	0.13
Total loss	35.	0.30

*kg/ha/yr x 0.89 = lb/ac/yr

In each of these studies, seepage losses of nitrogen were large compared to losses by surface runoff. On the other hand, phosphorus losses through surface runoff tended to be larger. However, phosphorus losses through seepage were sufficiently large to be of significance from a water quality standpoint.

Erickson and Ellis (1971) measured the nutrient content in drainage waters from three experimental farms in Michigan, and compared the nutrient losses to the amounts of fertilizer used.

	Nutrients added (kg/ha/yr)*		Nutrients lost (kg/ha/yr)*		Lost/added (percent)	
	N	P	NO_3-N	Tot-P	N	P
Ferden Farm	90	39	12	0.10	13	0.2
Davis Farm	39	50	8	0.09	20	0.2
Muck Farm	56	17	19	1.5	(34)	(8.8)

*kg/ha/yr x 0.89 = lb/ac/yr

Analyses of seepage from uncultivated, unfertilized land adjacent to the Muck Farm led to the conclusion that the high values of nutrient loss were in part due to accretion from surrounding lands, and that only a small part of the nitrogen and phosphorus added to the Muck Farm reached the drainage water.

Losses of nitrogen as a function of fertilizer usage was also studied by Broadbent and Chapman (1950). They grew vetch, clover and mustard in lysimeters at Riverside, California. The experiment covered a 15-year period and the average water application (rainfall plus irrigation) was 89 cm (39 inches).

	Fertilizer N applied (kg/ha/yr)*		
	0	112	224
	Nitrogen loss (kg/ha/yr)*		
Crop:			
Vetch	30	79	100
Clover	39	63	91
Mustard	20	34	45
Average loss for all crops and treatments:		56	

*kg/ha/yr x 0.89 = lb/ac/yr

Allison et al (1959) reported the results of experiments conducted near Columbia, South Carolina, in which crotalaria, millet, rye, cowpeas, and corn were grown in lysimeters. The lysimeters were 1.2 m(3.9 ft) deep and were filled with sandy soil. Fertilization was reported to be "low." The average annual rainfall was 108 cm (42.5 inches) during the 12-year period covered by the study.

	Nutrient loss (kg/ha/yr)*					
	NO_3-N		Total - N		Total - P	
	Range	Mean	Range	Mean	Range	Mean
Various crops	2.4-40	24	3.6-46	29	0.09-0.17	0.11
Fallow or crops returned to soil			38-140	90		
No crop:						
Fertilized		40		44		0.20
No Fertilizer		30		34		0.14

*kg/ha/yr x 0.89 = lb/ac/yr

Dreibelbis (1946) also found a correlation between nitrate loss and fertilizer usage, but Hendrick and Welsh (1938) reprted no significant differences in nitrate loss between fertilized and nonfertilized plots in a ten-year study conducted in England.

A summary of the data giving nutrient losses from croplands by seepage through the soil profile is given in Table 1. Based on considerations of nutrient pathways, data from lysimeter or subsurface drainage studies are probably most applicable for estimating nutrient loadings of lakes which receive irrigation return waters. The data may also be useful for estimating the flux of nutrients from croplands to groundwater aquifers, but subsequent transport would be highly speculative in most instances.

3.4.1.2 Nutrient Transport From Agricultural Lands In Surface Runoff

A separate data grouping was prepared for nutrient losses from agricultural lands by surface runoff. In studies of this type, samples of runoff water, including suspended matter, are collected periodically from fields or experimental plots. Runoff is not continuous, but occurs only when excessive water is applied through irrigation or rainfall.

Table 1 Nutrient Export From Croplands by Seepage Through Soil Profile

Crop--study	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	References
Corn--tile drainage, Lithuania	2.6					Kinderis (1970)
Corn-oats-hay rotation--lysimeter, 120 kg+ N/ha added, New York	43.					Bizzell (1944)
Corn-oats-hay rotation--lysimeter, New York	2.4					Bizzell & Lyon (1928)
Corn-oats-wheat-hay rotation--lysimeter, New York	5.4 with legumes 7.4 without legumes					" " "
Corn--lysimeter, Ohio	1.9					Dreibelbis (1946)
Corn--tile drainage, Ontario						
No fertilizer			6.1	0.20[a]		Bolton et al. (1970)
Fertilizer added			14.	0.26[a]		" " " "
Cotton--tile drainage, 280 kg N/ha added, California	4.1[b]					Meek et al. (1969)
Cotton--tile drainge, 220 kg N/ha and 36 kg P/ha added, California			13.		0.05	Johnston et al. (1965)
Cotton & rice--tile drainage, 300 kg N/ha and 52 kg P/ha added, California			110.		0.19	" " " "
Rice--tile drainage, 94 kg N/ha added, California			42.		0.60	" " " "

Table 1 (Continued) Nutrient Export From Croplands by Seepage Through Soil Profile

Crop--study	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	References
Oats--lysimeter, England	2.5					Hendrick & Welsh (1938)
Barley--lysimeter, England	8.					" " "
Wheat--lysimeter, Ohio	3.					Dreibelbis (1946)
Hay--lysimeter, England	9.					Hendrick & Welsh (1938)
Timothy--lysimeter, 140 kg N/ha added, New York	12.					Bizzell (1944)
Alfalfa--tile drainage, California			3.5		0.08	Johnston et al. (1965)
Alfalfa--tile drainage, Ontario			4.8	0.1		Bolton et al. (1970)
Alfalfa--lysimeter, Kentucky	11.					Karraker et al. (1950)
Lespedeza--lysimeter, Kentucky	65.					" " " "
Lespedeza & rye--lysimeter, Kentucky	17.					" " " "
Lespedeza & bluegrass--lysimeter, Kentucky	22.					" " " "
Legumes--tile drainage, Lithuania	1.5					Kinderis (1970)
Vetch--lysimeter, California			30.			Broadbent & Chapman (1950)
Clover--lysimeter, California			39.			" " "

Table 1 (Continued) Nutrient Export From Croplands by Seepage Through Soil Profile

Crop--study	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	References
Mustard- -lysimeter, California			20.			Broadbent & Chapman (1950)
Grasses--lysimeter, New York	2.8					Bizzell & Lyon (1928)
Grasses--tile drainage, Lithuania	0.3					Kinderis (1970)
Grasses & wheat--tile drainage, Lithuania	0.8					" "
Bluegrass sod--tile drainage, Ontario			0.3	0.01[a]		Bolton et al. (1970)
Meadow--lysimeter, Ohio	4.3					Driebelbis (1946)
Pasture--lysimeter, England	2.6					Hendrick & Welsh (1938)
Various crops--tile drainage, Idaho	35.			0.13		Carter et al. (1971)
Various crops--tile drainage, Washington	73.		100.	2.5	7.7	Sylvester & Seabloom (1962)
Not stated--tile drainge, Illinois	18.					Harmeson (1971)
Not stated--tile drainage, Michigan, Ferden Farm			12.		0.10	Erickson & Ellis (1971)
Davis Farm			8.		0.09	" " "
Muck Farm			19.		1.5	" " "

Table 1 (Continued) Nutrient Export From Croplands by Seepage Through Soil Profile

Crop--study	N (kg/ha/yr)*			P (kg/ha/yr)*		References
	NO_3	NH_4	Total	Dissolved inorganic	Total	
Fallow (no crop)--lysimeter, South Carolina	30.				0.14	Allison et al. (1959)
Fallow--lysimeter, New York	76.					Bizzell & Lyon (1928)
Fallow--lysimeter, Kentucky	82.					Karraker et al. (1950)

[a] Total dissolved P

[b] $NO_3 + NO_2$ for 8 mo.

*kg/h a/yr x 0.89 = lb/ac/yr

+kg x 2.2 = lb

As was the case for seepage studies, most surface runoff investigations were conducted to evaluate alternative farming practices, such as plowing techniques, crop rotations and fertilizer applications, from the standpoint of minimizing soil and nutrient losses from croplands. Consequently, particulate material was intentionally included in the samples, and particulate, along with dissolved, nutrients were measured in most instances. Study areas were often quite small and usually the areas were devoted to single crops. Fertilization and plowing were generally uniform within a study area, but large differences occurred between study areas.

Data from surface runoff studies are given in Table 2. The data are grouped according to crop, but other factors, such as slope, soil characteristics, farming practices, and antecedent soil moisture, as well as duration, frequency and intensity of precipitation, may have a greater influence on the quantity of nutrients lost from croplands. Slopes (not always given) ranged from 3-20% for the studies reported, and the annual amounts of rainfall and irrigation water ranged from 75 to 220 cm (30 to 87 inches).

Data from surface runoff studies may be useful for estimating nutrient inputs from agricultural lands immediately adjacent to lakes. However, as shown in Table 2, but far the largest amount of nitrogen and most of the phosphorus lost from croplands was associated with particulate matter. The likelihood that particulate matter will be transported sufficiently far to enter a lake must be taken into account before surface runoff data are used to estimate nutrient contributions from croplands which exist in distant portions of a watershed.

3.4.1.3 Nutrient Transport by Streams Draining Agricultural Lands

A number of studies are reported in the literature in which streams draining predominantly agricultural watershed were monitored for nutrient content. These studies typically involved the continuous measurement of streamflow and periodic sampling for nutrient determinations.

Table 2 Phosphorus and Nitrogen in Runoff from Crop Land

Land Use Practices			Solution			Particulate		Total		Reference
Crop	Fertilization	Management	Pi	NO_3-N	NH_4-N	Tot-P	Tot-N	P	N	
			kg/ha/yr*							
Row Crops										
Corn	M	G						1.90	4.4	Minshall et al. (1970)
"	H	P	0.07	0.80		0.11	12.6	0.18[2]	13.4[2]	Timmons et al. (1968)
"	H	P	0.70	1.20		0.04	4.2	0.74[2]	5.4[2]	Timmons et al. (1968)
"	M	P						1.50	23.0	Massey et al. (1953)
"	H	G	0.17[1]	1.70	1.30	1.05	36.6	1.22[2]	39.6[2]	Schuman et al. (1973)
"	M	G	0.11	1.00	0.90	0.58	23.1	0.69[2]	25.0[2]	Schuman et al. (1973)
"	H	G	0.05	0.20	0.30	0.08	2.6	0.13[2]	3.1[2]	Schuman et al. (1973)
"	H	G	0.13	1.41	0.33					Klausner et al. (1974)
"	H	P	0.49	2.46	0.99					Klausner et al. (1974)
"	M	G	0.05	0.39	0.14					Klausner et al. (1974)
"	M	P	0.16	0.40	0.21					Klausner et al. (1974)
"	M	P						6.60	182	Weidner et al. (1969)
Beans	H	G	0.18	1.46	0.40					Klausner et al. (1974)
Idle Crop Land										
Follow	H	-	0.05[1]	3.9	0.34		63	0.39[2]	66.9[2]	Timmons et al. (1968)
"	None	-			2.07					White & Williamson (1973)
Orchard and Vineyards										
Apple	M	G						1.40	0.9	Weidner et al. (1969)

Table 2 (Cont.) Phosphorus and Nitrogen in Runoff from Crop Land

Land Use Practices			Solution			Particulate		Total		Reference
Crop	Fertilization	Management	Pi	NO_3-N	NH_4-N	Tot-P	Tot-N	P	N	
			kg/ha/yr*							
Close-Grown Crops										
Oats	H	P	0.01[1]	0.75		0.03	5.2	0.04[2]	6.0[2]	Timmons et al. (1968)
"	M	G				0.32				White & Williamson (1973)
Wheat	M	G	0.04	2.46	0.21					Klausner et al. (1974)
"	M	P	0.09	2.29	0.61					Klausner et al. (1974)
"	H	G	0.17	0.93	0.42					Klausner et al. (1974)
"	H	P	0.32	1.14	0.36					Klausner et al. (1974)
"	M	G	0.21	0.50	1.30					Klausner et al. (1974)
"	M	P	0.08	0.60	0.15					Klausner et al. (1974)
"	M	P						0.90	24	Weidner et al. (1969)
Pastures and Meadows										
Hay	H	P	0.23[1]	3.5					3.5	Timmons et al. (1968)
Alfalfa	H	-						0.20	1.6	Johnston et al. (1965)
"	None	-				0.07				White & Williamson (1973)
Bromegrass	M	G	0.21	0.70	0.40	0.07	1.3	0.28[2]	2.4[2]	Schuman et al. (1973)

*kg/ha/yr x 0.89 = lb/ac/yr

[1]Total P in solution

[2]Total obtained as sum of inorganic nutrients in solution and particulate nutrient concentrations.

In many cases, sampling frequency was related to streamflow so that additional samples were collected during periods of high flow. The amounts of nutrients transported by the streams were then calculated from a streamflow record and a time series of nutrient concentrations. Two somewhat different approaches were used to accomplish this: 1) The flow hydrographs were subdivided into segments (usually centered about the dates on which water samples were collected), and a single nutrient concentration was then assumed to be characteristic of the total water mass passing the gaging station during each time segment; 2) sample analyses were used to develop concentration-stream-flow realtionships, and nutrient transport was computed by applying these realtionships to the stream hydrograph and integrating over appropriate time intervals.

A summary of nutrient transport from agricultural lands by steramflow is given in Table 3. Flux coefficients for these studies are less variable than for seepage or surface runoff studies, but reported values still cover a considerable range as shown below.

	Total N	Total P
	kg/ha/yr(lb/ac/yr)	
maximum	13.0(11.6)	2.3(2.0)
minimum	1.2(1.1)	0.03(0.03)
average	5.1(4.5)	0.38(0.34)

Of the 24 values of total-phosphorus that were compiled, only 7 were larger than 0.4 kg-P/ha/yr (0.36 lb/ac/yr), and 6 of the 7 were from the Midwest (2 from Illinois, 4 from Wisconsin). The largest value reported, 2.3 kg/ha/yr (2.05 lb/ac/yr), was determined from a study in Arkansas conducted by Gearheart (1969). In this study, streamflow draining a 3,072 km^2 (1,186 $mile^2$) watershed (80% agricultural, used primarily for pasture and poultry production) tributary to Beaver Reservoir was monitored for a seven-month period from October through April. Estimates of annual values were presented by the author. Whereas the phosphorus value was

Table 3 Nutrient Transport From Agricultural Watersheds By Streams

Location - comment	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	Tot-N / Tot-P	References
Illinois, Kaskaskia River basin							
Subbasin area / % agricultural							
32 km^{2+} / 86.0	11.				0.03[a]		Engelbrecht & Morgan (1959)[d]
320 " / 86.2					0.05[a]		
2,700 " / 81.7					0.85[a]		Harmeson et al. (1971)
5,100 " / 76.0					0.44[a]		
6,900 " / 68.9					0.11[a]		
13,500 " / 76.8					0.10[a]		
Connecticut, 85 km^2 watershed, 50% forested, "typical rural environment"			3.4		0.22	15.	Frink (1967)
Arkansas, 3072 km^2 watershed, 80% agricultural			3.6		2.3	1.5	Gearheart (1969)
Potomac River Basin (Catoctin Creek) 280 km^2, 80% farmland, 20% forest	3.8[b]		4.3		0.27	16.	Jaworski & Hetling (1970) Jaworski et al. (1969)
Ontario, tributaries of Bay of Quinte, 50% agricultural, 50% forests, many lakes and bogs							
River / Area (km^2)							
Trent / 13,000			2.1		0.11	19.	Johnston & Owen (1971)
Moira / 2,700			1.8		0.08	22.	
Salmon) / 1,660			2.4		0.07	34.	
Napanee)			3.0		0.14	21.	
North Carolina, Pigeon River watershed, 350 km^{2+}					0.17		Keup (1968)

Table 3 (Continued) Nutrient Transport From Agricultural Watersheds By Streams

Location - comment	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	Tot-N / Tot-P	References
Maine, rural areas, sparsely populated, Stetson R, 74 km^2			1.9		0.04	48.	Mackenthun et al. (1968)[d]
Wisconsin, average for 36 streams, base flow only, 5.7-370 km^2			1.2		0.10	12.	Minshall et al. (1969)
Ontario, Grand R watershed, 3500 km^2					0.07		Missingham (1967)[d]
Ontario, near Toronto							
River (Area (km^2))							
West Humber 130 (dairy farms)			3.2		0.21	15.	Owen & Johnson (1966)
Little Rouge 78 (mixed farms)			8.4		0.35	24.	Neil, Johnson & Owen (1967)
Antona 54 (mixed farms)			4.0		0.17	24.	
England							
Arable land			13.				Owens (1970)
Permanent pasture			8.				
Wisconsin, tributaries to Lakes:							
Monona	← 4.9	→	6.7	0.06			Sawyer (1947)
Waubesa	← 5.5	→	7.6	0.11	0.44	17.	
Kegonsa	← 7.2	→	9.2	0.11	0.46	20.	
Prince Edward Island, 26 km^{2+} watershed, 28% potato fields, remainder in pasture & woodlot				0.21			Smith (1959)

Table 3 (Continued) Nutrient Transport From Agricultural Watersheds by Streams

Location - comment	N (kg/ha/yr)* NO_3	NH_4	Total	P (kg/ha/yr)* Dissolved inorganic	Total	Tot-N / Tot-P	References
Ohio, 123 ha watershed, 25% woodlots, 50% pasture, 25% cropland (data for 4 consecutive years)	1.1		1.7	0.03			Taylor et al. (1971)
	2.2		3.1	0.08			
	9.1		10.6	0.07			
	1.8		4.4	0.08			
Wisconsin, Menomonee R watershed					1.6[c]		Zanoni (1970)[e]
Wisconsin, 546 ha watershed, dairy farming, 0.15% slopes	3.1[b]	1.3	8.8	0.58	0.77	11.	Zitter (1968)

[a] Ortho-P + maximum inorganic condensed-P - Authors state that total-P values may be 20-30% higher than those reported.

[b] $NO_3 + NO_2$

[c] Total dissolved - Author states that values are within a few percent of total-P.

[d] Data given as loss/day or loss for part of a year--extrapolated linearly to an annual value.

[e] Approximated from data presented.

* kg/ha/yr x 0.89 = lb/ac/yr

+ km^2 = 247 ac.

larger, the flux of nitrogen, 3.6 kg-N/ha/yr (3.2 lb/ac/yr), was less than the average for other data included in this group, and the ratio of Tot-N/Tot-P was only 1.5. Ratios for all other studies were greater than 10.

Phosphorus transport in the Kaskaskia River watershed in Illinois was reported by Engelbrecht and Morgan (1959, 1961). Water samples were collected at approximately weekly intervals at four sites and monthly at three additional locations during the period from April-December 1956. Some municipal effluents were discharged to the river, but contributions from these sources were subtracted from the total load, and values for land drainage are presented by the authors. Results presented as phosphorus loss/unit area/day were extrapolated linearly to annual values here.

Mackenthun, Keup, and Stewart (1968) and Mackenthun (Chairman, 1966) reported studies on the results of tributaries to Lake Sebasticook, Maine. The streams studied drained sparsely populated rural areas with no significant waste discharges. The primary crops were potatoes, apples, alfalfa, beans, and corn which received an average of 82 kg/ha (73 lb/ac) of phosphorus as fertilizer. Precipitation was 102 cm/yr (40 inches/yr). Water samples were collected during one- to two-week periods in February, May, July-August, and October-November. Nutrient losses per day were reported by the authors for each of the four sampling periods.

	Nutrient Loads (kg/ha/day)*			
	Stetson Stream		Mulligan Stream	
	N	P	N	P
Winter	0.0029	0.0	No Flow	
Spring	0.0086	0.00015	.00084	.00003
Summer	0.0083	0.00012	No Flow	
Fall	0.0015	0.00018	.0013	.00005

*kg/ha/day x 0.89 = lb/ac/day

Annual values were computed here by multiplying each daily load per season by 91.25 and summing.

Phosphorus transport from a 3,500 km^2 (1,350 $mile^2$) agricultural watershed in Ontario (Grand River) was repoted by Missingham (1967).

Month	Total P (kg/ha/day)*
December	0.00014
January	0.00025
February	0.00021
Average	0.00020

*kg/ha/day x 0.89 = lb/ac/day

An average value of 0.0002 kg/ha/day (0.00018 lb/ac/day) converts to 0.07 kg/ha/yr (0.062 lb/ac/yr) which is, most likely, an underestimate of the amount of phosphorus transported annually from the basin.

Minshall, Nichols, and Witzel (1969) carried out a two-year study to determine the amount of nutrients in base flow of southwestern Wisconsin streams. Flow rates were determined for 36 streams with drainage areas varying from 570 to 37,000 ha (1,410 to 91,430 ac). Samples were collected, and flows were measured at times when no surface runoff was entering the streams.

The area studied was 90% agricultural, with 40% in contour strip-cropped farmland (corn, oats and alfalfa), 40% in pasture, and 10% woodland. Livestock enterprises were prevalent. Soils were moderately and well-drained silt loams, and the mean annual precipitation for the area was 83 cm (32.7 inches). An average of 9 kg/ha/yr (8 lb/ac/yr) nitrogen was applied as manure or artificial fertilizers.

	Nutrient Loss (kg/ha/yr)*			
	1966		1967	
	Tot-N	Tot-P	Tot-N	Tot-P
High	4.2	0.25	5.5	0.49
Low	0.4	0.01	0.5	0.03
Weighted Average	1.1	0.08	1.4	0.12

*kg/ha/yr x 0.89 = lb/ac/yr

Zanoni (1970) conducted a study of the Menomonee River basin in southeastern Wisconsin, and reported that an average of 1.18 kg/ha (1.05 lb/ac/yr) of total soluble phosphorus was contributed annually to Lake Michigan from land drainage in the watershed. The total watershed contains 350 km^2 (135 $mile^2$) of which 38% is agricultural, 43% is urban and the remaining 19% is woodlots, parks and unproductive land. An analysis of runoff from sub-basins in the watershed showed that urban areas contributed 0.58 kg/ha/yr (0.52 lb/ac/yr). Drainage from the remainder of the watershed, primarily rural lands, can be computed to yield a contribution of 1.63 kg/ha/yr (1.45 lb/ac/yr).

3.4.2 Nutrient Loadings for the Great Lakes

Estimates have been made of the P loadings for the five Great Lakes and of the N loadings for Lake Huron (Zar, 1972; Great Lakes Water Quality Board, 1973). The greater emphasis on P than N in these estimates apparently reflects the focus on P in regulatory programs planned or implemented to reduce the input of nutrients into the Great Lakes (Zar, 1972; Great Lakes Water Quality Board, 1973).

The P loading for Lake Michigan was estimated by the Phosphorus Technical Committee to the Lake Michigan Enforcement Conference (Zar, 1972). These estimates are summarized in Table 4. For direct and indirect point sources of P discharge, the estimates were based on the assumptions of a P concentration of 10 mg/l in the influent of sewage treatment plants, a discharge to domestic waste water of 1.64 kg (3.65 lb) of P per person per year and 454 liters (120 gallons) per person per day, or were calculated from actual effluent data where available. Consequently, the values for point sources reflect mainly municipal rather than industrial wastes. Direct point sources are wastes discharged directly into the lake, while indirect point sources are wastes discharged into tributaries to the lake. Diffuse sources estimates were based in part on soil erosion (0.5 kg P/metric ton (1 lb P/ton) of sediment or 0.9×10^6 kg P/yr (2×10^6 lb P/yr for the lake), and on other sources (decaying leaves and crop residues; dissolved nutrients in runoff). Details on the basis for estimates of the latter sources were not given. However, estimated total input from diffuse sources

ranged from 12. to 30 kg/km^2/yr (1 to 25 lb/mile^2/yr) or 0.5 to 3.2 x 10^6 kg/yr (1 to 7 x 10^6 lb/yr) basin wide.

Estimates of P loadings from other than direct point sources were also made based on measurements of P concentrations and flow for tributaries to Lake Michigan during 1969. The total input from tributaries and direct point sources (7,764 metric tons/yr or 8,560 tons/yr) is approximately equal to the total input based on estimates (explained previously) of indirect point sources and diffuse sources (7,809 metric tons/yr or 8,609 tons/yr). The close agreement between the two estimates tends to support their validity (Table 4).

The P loadings estimated for Lake Michigan (Table 4) indicate that the contribution from diffuse sources (including agricultural lands) is appreciable (about 25% of the total). However, loadings for diffuse sources are variable and more difficult to estimate than for point sources. Furthermore, the specific contribution from agricultural lands was not estimated. In spite of these problems in assessment, it appears that the P contribution from agricultural lands is significant and will increase in proportion as the input from point sources is reduced on the basis of the requirement that all municipalities achieve 80% reduction of total P entering waste water treatment plants that discharge to Lake Michigan or to one of its tributaries (Zar, 1972).

The P loading estimates for the five Great Lakes are shown in Table 5 according to the contributions from tributaries, industrial sources and municipal sources. The values for municipal and industrial sources are based on direct discharges into the lakes, while tributary loading include diffuse sources (including agricultural lands) and point discharges along the tributaries (EPA, 1972; Great Lakes Water Quality Board, 1973)[1].

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[1] Data for nutrient budgets was taken from Section III-C, Water Quality Conditions (Great Lakes Water Quality Board, 1973). Discrepancies exist in some cases between these values and values in Section IV-C, Eutrophication, of the same publication.

Table 4. Phosphorus Loadings to Lake Michigan (Zar; 1969)[1]

Source	Amount	
	metric tons/year	(tons/year)
Direct point sources	1,771	(1,953)
Indirect point sources	4,221	(4,654)
Sub-total	5,993	(6,607)
Diffuse sources	1,816	(2,002)
Total (Based on estimates of sources)	7,809	(8,609)
Tributaries (1969)	5,993	(6,607)
Direct municipal and industrial sources	1,771	(1,953)
Total (Based on measurement of tributary inputs)	7,764	(8,560)

[1]Estimates do not include direct atmospheric fallout and precipitation and combined sewer overflows.

Table 5.

Total Phosphorus Loadings to the Great Lakes From Tributaries, Industrial and Municipal Sources

(Great Lakes Water Quality Board, 1973)

	Superior		Michigan[1]		Huron		Erie		Ontario	
Canada	metric tons/year						(tons/year)			
Tributaries	608	(670)			1,143	(1,259)	934	(1,030)	1,705	(1,880)
Industrial	118	(130)			1	(1)	54	(60)	299	(330)
Municipal	95	(105)			77	(85)	36	(40)	2,413	(2,660)
United States										
Tributaries	980	(1,080)	5,986	(6,600)	1,390	(1,531)	4,185	(4,615)	1,977	(2,180)
Industrial	-	-	-	-	nd	(nd)	209	(230)	-	-
Municipal	363	(400)	1,814[2]	(2,000)	4	(4)	1,900	(2,095)	1,224	(1,350)
	2,164	(2,385)	7,800	(8,600)	2,615	(2,880)	7,318	(8,070)	7,618	(8,400)
Input from other Great Lake(s)	-	-	nd	(nd)	885[3] 231[4]	(975) (255)	9,578[5]	(10,560)	5,505[6]	(6,070)
	2,164	(2,385)	7,800	(8,600)	3,731	(4,110)	16,896	(18,630)	13,123	(14,470)

[1]Based on estimates reported by Zar (1972)

[2]Includes industrial and municipal sources

[3]Lake Superior, including St. Mary's River

[4]Lake Michigan

[5]Lake Huron, Lake St. Clair, and the Detroit River

[6]Lake Erie, including the Niagara River and Welland Canal

Table 6.
Nitrogen Sources of Lake Huron
(Great Lakes Water Quality Board, 1973).

Source	Amount	
	metric tons/year	(tons/year)
Canada		
Tributaries	16870	(18600)
Industrial	ns	(ns)
Municipal	181	(200)
	17,051	(18,800)
United States		
Tributaries	nd	
Industrial		
Municipal		
	17,051	(18,800)
Input from Lake Superior including St. Mary's River	19,410	(21,400)
Input from Lake Michigan	19,682	(21,700)
	56,143	(61,900)

Estimates for the Great Lakes show that diffuse and indirect point sources account for a major part of the P loadings. The proportion of the total P loading contributed by tributaries ranges from about 77% for Lake Michigan to 28% for Lake Ontario. The lower proportions for Ontario and Erie reflect the importance of contributions from the preceding lake in the chain. While the estimates for the Great Lakes show the importance of P contained in tributaries, these values do not show the relative importance of diffuse sources (including agricultural land) and point waste discharges along the tributaries. Municipal waste discharges into tributaries likely account for a large part of the total tributary load. More detailed information is needed to assess the agricultural contribution.

Apparently, loadings of N to the Great Lakes have received considerably less attention than estimates of P loadings. Estimates were reported for the N loading to Lake Huron from the Canadian side by the Great Lakes Water Quality Board (1973) as shown in Table 6. In contrast to the P loading (about 60% from direct municipal and industrial sources for the Canadian side), the N was estimated to arise almost completely from tributary input and from Lakes Superior and Michigan.

3.4.3 Estimation of the Contribution of Agricultural Lands to the Nutrient Loading for the Great Lakes

Information on the amounts of nutrients transported from agricultural land in surface runoff, summarized in section 3.4.1.2, provides a basis for estimating the amounts of nutrients transported to the Great Lakes from agricultural lands. Use of data obtained from small watersheds or field plots allows estimation of nutrient loadings according to the type of agricultural land use. However, it must be recognized that considerable variability exists in data available for a given land use category, e.g., row crops, (see section 3.4.1.2) and that estimates made on this basis are only approximations at best.

The data on rates of nutrient transport in surface runoff (section 3.4.1.2) was averaged to obtain typical nutrient loading rates for each land use category. Values were estimated for inorganic phosphate in solution (soln Pi), total phosphorus, inorganic nitrogen (NH_4-N+NO_3-N) in solution (soln N) and total nitrogen (Tables 7 & 8). Input rates were obtained by averaging the data for each land use category (Table 11, section 3.4.1.2). Data on agricultural land use was taken from the Great Lakes Basin Framework Study, Appendix 13 (1971).

The estimated phosphorus loading to the Great Lakes from agricultural lands was 1830 metric tons/year (2020 tons/year) of inorganic P in solution and 12,000 metric tons/year (13,300 tons/year) for total P. These values correspond to 9 and 60 percent, respectively, of the total P loading for the Great Lakes estimated by the Great Lakes Water Quality Board (1973) and shown in Table 5, section 3.4.2. Loadings for N were 27,800 metric tons/year (30,600 tons/year) for inorganic N in solution and 399,000 metric tons/year (439,000 tons/year) for total N. As discussed below, total N and P values likely are overestimates, as part of the soil eroded from small agricultural plots is deposited as sediment prior to reaching the Great Lakes. Furthermore, some of the particulate N and P transported to the Great Lakes is unavailable and/or settles to the lake bottom before utilization by aquatic plants can occur (see Section 3.5).

The input rates for inorganic P in solution (Table 7) were similar (about 0.2 kg/ha/yr or 0.18 lb/ac/yr) for the different crop use categories, with the exception of idle crop land. Limited data was available for the latter category. These values are partly affected by the amount of data available and the area in which the data was collected. Considering the range in data used to obtain the input rates, the differences between crop use categories are not significantly different for solution P. This suggests that crop use may not be the best approach for classification of agricultural land for estimating nutrient loading. A greater range was obtained in input rates for total P. The differences reflect the influence of crop cover on the land surface on soil erosion. Thus, the highest rate was obtained for row crops. The values for idle crop land and

Table 7. Estimated Annual Loadings of Phosphorus to the Great Lakes by Runoff from Crop Land (U.S. Region only)

Crop	Area	Input Rate		Annual Loadings	
		Soln P_i	Total P	Soln P_i	Total P
	1,000 hectares (1,000 acres)	kg/ha/yr (lb/ac/yr)		metric tons/yr (tons/yr)	
Row Crops	3,949 (9,750)	0.21 (0.19)	1.62 (1.45)	829 (914)	6,400 (7,050)
Close-Grown Crops	1,458 (3,600)	0.13 (0.12)	0.47 (0.42)	190 (209)	685 (755)
Pastures and Meadows	2,719 (6,715)	0.22 (0.19)	0.24 (0.21)	598 (659)	653 (720)
Orchards and Vineyards	243 (600)	0.21[1] (0.19)	1.40 (1.25)	51 (56)	340 (375)
Idle Crop Land	3,219 (7,947)	0.05 (0.04)	1.23 (1.10)	161 (178)	3,960 (4,370)
Total	11,588 (28,612)			1,829 (2,017)	12,038 (13,270)

[1] Assumed to be the same as row crops.

Table 8. Estimated Annual Loadings of Nitrogen to the Great Lakes by Runoff From Crop Land (U.S. Region only)

Crop	Area	Input Rate		Annual Loadings	
		Solution N	Total N	Solution N	Total N
	1,000 hectares (1,000 acres)	kg/ha/yr (lb/ac/yr)		metric tons/yr (tons/yr)	
Row Crops	3,949 (9,750)	1.56 (1.39)	37.0 (33.0)	6,160 (6,780)	146,000 (161,000)
Close-Grown Crops	1,458 (3,600)	1.67 (1.49)	15.0 (13.4)	2,440 (2,680)	21,900 (24,100)
Pastures and Meadows	2,719 (6,715)	2.30 (2.05)	2.50 (2.23)	6,250 (6,880)	6,800 (7,480)
Orchards and Vineyards	243 (600)	1.56[1] (1.39)	37.0[1] (33.0)	379 (417)	8,990 (9,890)
Idle Crop Land	3,219 (7,947)	3.90 (3.48)	66.9 (59.7)	12,600 (13,800)	215,000 (237,000)
Total	11,588 (28,612)			27,829 (30,557)	398,690 (439,470)

[1] Assumed to be the same as row crops.

orchards reflect data obtained from fallow soil and cultivated orchards, respectively. This may not be typical of land in these categories in the Great Lakes Basin.

As with P, input rates for inorganic N in solution were generally similar for the different crop categories and were low compared to total N. The rates for total N reflected the importance of soil erosion on bare soil.

While data for runoff collected from small plots provides information related to agricultural crop cateogry, these data do not reflect the processes, such as sedimentation, which occur during transport of the water to the Great Lakes. On this basis, data on streams draining agricultural lands probably provides a better measure of nutrient transported to lakes. Even using stream data, the calculated total nutrient loadings will over-estimate the available nutrient loadings (see Section 3.5).

Considerable data is available on nutrients in streams draining agricultural lands (see Section 3.4.1.3). This data can be used to estimate typical input rates for agricultural lands but can not be used to estimate differences for the various crop use categories as the streams generally draining areas including several agricultural land uses. The stream data in Table 3 (Section 3.4.1.3) was used to calculate input rates for dissolved inorganic P, total P, dissolved inorganic N, and total N (Table 9). The value for dissolved inorganic P is slightly higher than the value obtained from runoff data, 1,300 metric tons/year (1,400 tons/year as compared to 1,830 metric tons/year (2,020 tons/year). However, the total P loading rate based on stream data, 4,200 metric tons/year (4,600 tons/year), was considerably lower than the loading rate based on runoff data, 12,000 metric tons/year (13,300 tons/year). The agricultural land loading to the Great Lakes, based on data from streams draining agricultural land is 21 percent of the total loading (Table 9) and represents a better estimate than the value based on runoff.

In the case of N, stream data gives similar loading rates for inorganic N in solution and total N. This reflects the high proportion of

Table 9. Estimated Annual Loadings of Nutrients to the Great Lakes based on Stream data (U.S. Region Only)

Total Cropping Area = $11,588 \times 10^3$ hectares
($28,612 \times 10^3$ acres)

Nutrients	Input Rate	Annual Loadings	Proportion of Loading to the Great Lakes	
			Tributary P	Total P
	kg/ha/yr (lb/ac/yr)	metric tons/yr (tons/year)	%	
Phosphorus				
Dissolved Inorganic P	0.11 (0.10)	1,280 (1,400)	9	6
Total P	0.36 (0.32)	4,170 (4,590)	29	21
Nitrogen				
Inorganic N	3.80 (3.39)	44,000 (48,400)		
Total N	4.90 (4.38)	56,800 (62,500)		

Table 10. Comparison of Phosphorus Sources for Lake Erie and Lake Ontario as Percent Contribution

Inputs	Lake Erie[1]	Lake Ontario[2]
	%	
Direct Precipitation	5	-
Land Runoff		9
Urban	4	
Agricultural or Rural	15	
Forest	-	
Wastewater		
Municipal	73	86
Industrial(Direct)	4	5

[1]Weibel, 1969. Excludes input from Lake Huron which was 12.6% of the total input.

[2]EPA, 1971. Excludes input from Lake Erie.

NO_3-N in the data used (Table 3). As with P, the total N loading rate is considerably lower when based on stream data rather than on runoff data. The total N loading from agricultural land of 56,800 metric tons/year (62,500 tons/year) based on stream data is considered a more reliable estimate than the value of 399,000 metric tons/year (439,000 tons/year) based on runoff. Insufficient information is available on the total N loading to the Great Lakes to compute the proportion arising from agricultural lands.

The proportion of the P loading to the Great Lakes estimated to originate from agricultural lands (21%) can be compared to estimates obtained by others using different approaches. In the case of Lake Michigan (see Table 4), it was estimated that about 23% of the total loading was from diffuse sources. This would include urban runoff and other non-point sources. Out of the total P loadings to Lake Erie, about 14% was estimated to come from agricultural land, while 9% was contributed to Lake Ontario by runoff which also included urban runoff (Table 10).

In summary, agricultural land is estimated to contribute about 20 percent of the total P loading to the Great Lakes and about 30 percent of that contributed by tributaries to the Great Lakes. This proportion will increase as treatment of municipal wastes is accomplished. The largest proportion likely arises from row crops, but appreciable amounts originate from other crop cateogries. Within the present accuracy of loading rates for different land use categories, the differences in loading rates are primarily a function of the land areas in the respective categories rather than differences in loading rates.

3.5 Mobility of N and P within Soil-Water, Ground Water and Surface Water Systems.[1]

[1] The parts of this section dealing with P are based on the review by Ryden *et al.* (1973).

3.5.1 Introduction

The transport of P and N from agricultural lands to surface waters is a function of the physical and chemical factors controlling the mobility of P and N in soils. An understanding of these factors is essential to evaluating the extent of P and N transport in surface, subsurface and ground water runoff as related to land use activities and to developing management programs to minimize nutrient transport.

This section deals with the factors controlling P and N mobility in soil-water systems. Quantitative aspects P and N transport in runoff waters are considered in section 3.4.

3.5.2 Terminology

3.5.2.1 Hydrology and P and N Sources

"Watershed" (drainage basin, catchment area) -- A part of the surface of the earth that is occupied by a drainage system, which consists of a surface stream, or a body of standing (impounded) surface water, together with all tributary surface streams and bodies of standing surface water.

"Stream" -- A general term for a body of flowing water. In hydrology the term is usually applied to the water flowing in a natural channel.

"Stream flow" -- The discharge (of water) that occurs in a natural channel.

"Runoff" -- That part of precipitation that falls on land and ultimately appears in surface streams and lakes. Runoff may be classified further according to its source.

"Surface runoff" (overland flow) -- The part of rainwater or snowmelt which flows over the land surface to stream channels. Surface runoff may also enter standing waters directly or be consolidated into

·tificial channels, e.g., storm sewers in urban areas ("urban runoff"), :fore entering a stream or body of standing water.

"Subsurface runoff" (storm seepage) -- That part of precipitation ıich infiltrates the surface soil and moves towards streams as ephemeral, ıallow, perched ground water above the main ground-water level. In many gricultural areas subsurface runoff may be intercepted by artificial rainage systems, e.g., tile drains, accelerating its movement to streams.

"Ground water runoff" (base runoff) -- That part of precipitation hat has passed into the ground, has become ground water, and is subsequently ischarged into a stream channel or lake as spring or seepage water.

In addition to runoff the other potential contributors to ;treams and standing waters are precipitation incident on the water ;urface and industrial and sewage effluents.

McCarty (1967) and Vollenweider (1968) have made a useful division of sources of P to surface waters based on the ease of quantification. Point sources enter at discrete and identifiable locations and are therefore amenable to direct quantification and measurement of their impact on the receiving water. Major point sources include effluents from industrial and sewage-treatment plants. Diffuse sources may be defined as those which at present can only be partially estimated on a quantitative basis and which are probably only amenable to attenuation rather than elimination. Diffuse sources require the most investigative attention. Vollenweider (1968) further divided diffuse sources into:

i) Natural sources such as eolian loading, and eroded material from virgin lands, mountains and forests.
ii) Artificial or semi-artificial sources which are directly related to human activities such as fertilizers, eroded soil materials from agricultural and urban areas, and wastes from intensive animal rearing operations.

The loads of P imparted to runoff and streams from natural diffuse sources provide a datum line against which the magniture of P loads from artificial sources may be compared.

3.5.2.2 Forms of P and N

In natural systems, P occurs as the orthophosphate anion (PO_4^{3-}) which may exist in purely inorganic form ($H_2PO_4^-$ and HPO_4^{2-}) or be incorporated into an organic species (organic P). Under certain circumstances inorganic orthophosphate may exist as a poly- or condensed phosphate. A secondary distinction is made between particulate and dissolved forms of P, the split conventionally being made at 0.45 μm.

Other terminology used is as follows:

"Total P" -- all forms of P in a runoff or stream sample (dissolved and particulates in suspension) as measured by an acid-oxidation treatment (e.g., acid ammonium persulfate).

"Dissolved inorganic P" -- P in the filtrate after 0.45 μm separation determined by an analytical procedure for inorganic orthophosphate.

"Organic P" -- may be determined within the dissolved and particulate fractions by the difference between the total P and inorganic P.

In soils and waters, N occurs as the nitrate ion (NO_3^-), the ammonium ion (NH_4^+) and in organic compounds (amino N, heterocyclic N, etc.). Small amounts of nitrite (NO_2^-) may be present but the amounts are usually small compared to other forms and NO_2^- is generally not measured in runoff. All of these forms may occur in the dissolved or particulate fraction. The conventional division between the dissolved and particulate fractions is based on filtration using a 0.45 μm pore size filter.

"Total N" - all forms of N in the sample (dissolved and particulate), including NH_4-N, NO_3-N and organic N.

"Dissolved inorganic N" - the sum of NO_3-N and NH_4-N in the filtrate after 0.45 μ m separation. NO_3-N and NH_4-N are determined by separate methods and may be expressed as such. NO_3-N is associated mainly with the dissolved phase and the necessity of removing fine particulates to avoid overestimations is less critical than with P and NH_4-N.

"Organic N" - is determined as the NH_4-N formed on Kjeldahl digestion of the dissolved or particulate fraction.

3.5.3 Physical Factors Affecting the Mobility of P and N

All terrestrially-derived diffuse sources of N and P are associated with the movement of water in contact with a solid phase. The solid phase may be stationary with respect to water flow, or may move in the flow at some speed equal to or less than the flow. Precipitation disposed of as subsurface or ground water runoff is primarily in contact with a stationary solid phase, namely the soil profile and, in the case of ground-water runoff, possibly the bed rock. Consequently, the amounts and concentrations of N and P carried in subsurface and ground-water runoff will be influenced by the time of contact with any component in the soil profile capable of interacting with N and P dissolved in the percolating water and by the concentrations of dissolved N and P that the soil components maintain in the soil solution. Time of contact between the percolating solution and any soil component will in turn depend on the rates of infiltration and percolation into the soil.

Some of the theories developed to describe water movement in soils can be applied to evaluate the potential loss of N and P from various soil types as a result of subsurface runoff. Gardner (1965) developed equations to describe the movement of nitrate in the soil profile due to leaching. The chemical interactions that occur between dissolved inorganic N and P and soil components (discussed later), when water percolates through the soil, must be taken into consideration. Inclusion of a terms in the equations developed by Gardner (1965) to describe the relationships between N and P in particulate and aqueous phases is therefore necessary. This could take the form of a linear adsorption isotherms relevant to the concentrations of dissolved inorganic N and P maintained in the solution of a particular soil. Biggar and Corey (1969) have also reviewed the literature on infiltration and percolation of water in agricultural soils as it pertains to nutrient movement.

The movement of solid phase material in contact with natural waters occurs during surface runoff and in streams. The amount of solid material capable of entering surface runoff will depend on the intensity of rainfall, physical and chemical attachment between various solid components, and the amounts of energy of runoff waters (Guy, 1970). It is the energy of surface runoff or stream water, however, that governs the amounts of a specific size fraction of particulate materials which will remain in suspension during water flow.

The primary source of particulate material to surface runoff and streams is eroding soil (Guy and Ferguson, 1970), although in urban areas with little ongoing development, particulates may be dominated by specifically urban detrital material (e.g. street litter and dust) and organics derived from urban vegetation.

The total on-site losses of soil due to sheet and rill erosion are not necessarily delivered to streams. The amount of sediment that travels from a point of erosion to another point in the watershed is termed the sediment yield (Johnson and Moldenhauer, 1970). Consequently the Universal Soil Loss Equation used to predict field soil losses on an average annual basis (Wischmeier and Smith, 1965) must be corrected when used to predict sediment loads in streams because deposition of particulates may occur on the land surface as a result of slope variations before surface runoff reaches a stream. It is for this reason that estimates of soil loss in surface runoff from sites within a particular watershed cannot be translated into total N and P losses through a knowledge of the total N and P contents of the soil, if the N and P losses are to be related to P enrichment of surface waters.

An associated complication arises from the fact that soil P is primarily associated with the solid phase. In contrast, NO_3-N is soluble in soil and moves in solution in runoff. As soil erosion is a selective process with respect to particulate size, selectivity has been observed for P loss in surface runoff. The extent of the selectivity depends on the particle sizes with which most of the soil P is associated.

This observation has lead to the concept of enrichment ratios (E.R.), which for P are calculated as the ratio of the concentration of P in the particulate phase of surface runoff to the concentration of P in the source of the particulate phase. This effect was first considered by Rogers (1941) who observed E.R. values of 1.3 for total P and 3.3 for "0.002N H_2SO_4 extractable" P for a silt loam situated on a 20-25% slope. Other values range from 1.5 to 3.1 for total P (Knoblauch et al., 1942; Neal, 1944; Stoltenberg and White, 1953), whereas Massey and Jackson (1952) observed values between 1.9 and 2.2 for "water-soluble plus pH extractable" P for silt loams in Wisconsin. The selective nature of surface runoff with respect to P is due to selective removal of fine soil particulates as a result of the energy limitations of runoff and the fact that a large percentage of total soil P is frequently associated with clay-sized material. A high proportion of the total P in soils may be associated with the clay-sized fraction (Scarseth and Chandler, 1938; Williams and Saunders, 1965; Syers et al., 1969). Greater selectivity of fines and consequently particulate P will occur as the energy of surface runoff decreases. Stoltenberg and White (1953) observed that as precipitation disposed of as surface runoff decreased from 7 cm/hr (2.75 inches/hr) to 0.025 cm/hr (0.01 inches/hr), the clay content of eroded material from a soil with a clay content of 16-18% increased from 25 to 60%. This had obvious implications in relation to the nature of the sediment load carried by a stream and the interactions of P between the solid and aqueous phases, particularly during periods of surface runoff. It should be pointed out, however, that although the P content of the sediment load may increase as surface runoff diminishes, as may be predicted from the work of Stoltenberg and White (1953), the total P load may not change, and could decrease, due to lower sediment loads.

The particulate material carried in streams may be divided into bed load and wash or suspended load. The bed load, which may also have a contribution from existing stream sediment, is that which moves along or close to the stream bed, whereas the wash load is maintained in the flow by turbulence (Johnson and Moldenhauer, 1970). By inference from

the selectivity of surface runoff for fine soil particulates, the wash load will be high during surface runoff events. Furthermore, Johnson and Moldenhauer (1970) suggest that the wash load travels at about the same velocity as the water with which it is in contact. Consequently, P assoicated with the clay- and silt-sized particulates constituting the wash load will move between any two points in the stream profile at the same speed as the ambient dissolved forms of P.

Increased turbulence in streams during high flow, or arising from an increasing gradient, will tend to maintain in suspension particle sizes more characteristic of the bed load, and may even resuspend existing stream bed sediment. In a study of total P loads in the Pigeon River, N.C., Keup (1968) noted that an increase in gradient from 2.81 to 4.35 m/km, over which no tributaries entered the main stream, resulted in a 90.8 kg/day (200 lb/day) increase in the total P load carried.

It appears that in the majority of cases a large proportion of particulate N and P in streams arises from soil erosion. The N and P may be stored in stream bed sediments, but unless the stream is actively aggrading, the amount stored will be less than the inflow (Keup, 1968). That which is stored is liable to resuspension and transport due to turbulence during periods of high flow.

3.5.4 Chemical Factors Affecting the Mobility of P

3.5.4.1 Nature of Soil P

Whenever water containing a particular concentration of dissolved inorganic P comes into contact with soil material, there is a possibility for sorption, desorption, or dissolution reactions to take place. The types of reactions are the same regardless of whether they occur under conditions existing in the soil profile, surface runoff, or streams, all of which may be regarded as soil-water ecosystems. Although in some cases biological assimilation may initially affect the distribution of P between dissolved and particulate phases of soil-water ecosystems, the

distribution of P between these phases will be determined by the nature of the inorganic particulates and the concentrations of dissolved inorganic P in solution (Keup, 1968; McKee et al., 1970; Ryden et al., 1972b).

On the basis of solubility product criteria, it has been postulated that discrete phase crystalline Fe and Al phosphates exist in soils (Hemwall, 1957; Kittrick and Jackson, 1956; Chakravart and Talibudeen, 1962). The general occurrence of discrete Fe and Al phosphates seems doubtful based on the ion product data presented by Bache (1964) and the experimental observations of Hsu (1964). That Fe and Al phosphates are formed as a temporary phase in the vicinity of phosphate fertilizer particles due to conditions of localized high acidity and P concentration is well established (Lindsay and Stephenson, 1959; Huffman, 1969). The compounds, however, will not be stable as the dissolved inorganic P concentration in the soil solution or aqueous portion of other soil-water ecosystems decreases.

The calcium phosphate minerals, apatite (Shipp and Matelski, 1960) and calcic fertilizer-soil reaction products (Huffman, 1969) have been identified in soils. The amounts of apatite are only appreciable in weakly weathered soils (Williams et al., 1969), as predicted by the weathering indices of Jackson (1969). Calcic fertilizer-soil reaction products may be present in neutral and calcareous surface soil horizons, and their importance in maintaining high concentrations of dissolved inorganic P in soil-water ecosystems should not be overlooked.

It has been demonstrated that the uptake or sorption of P from solution by soils is significantly related to the presence of short-range-order (amorphous) oxides and hydrous oxides of Fe and Al (Williams et al., 1958; Gorbunov et al., 1961; Bromfield, 1965; Hsu, 1964; Saunders, 1965). Furthermore, "pure" oxides and hydrous oxides of Fe and Al, and short-range-order aluminosilicates have also been shown to be particularly effective in the sorption of inorganic P from solution (Gastuche et al., 1963; Muljadi et al., 1966; Hingston et al., 1969).

The sorption of inorganic P by Fe and Al oxides and hydrous oxides is known to be rapid, as is the sorption of P by soils. Furthermore, short-range-order Fe and Al oxides and hydrous oxides are ubiquitous in soils (Hsu, 1964), their relative amounts depending on parent material, climatic, and drainage conditions, and occur mainly as coatings on other soil components. Shen and Rich (1962) and Jackson (1963) have noted the occurrence of Al hydroxypolymers, and Dion (1944) and Roth et al. (1969) have reported the presence of Fe oxide and hydrous oxide coatings on clay mineral surfaces. Such coatings, in conjunction with the greater surface area of the clay fraction compared to that of the other particle size fractions 50% of the total P in soils may be assoicated with the clay fraction, as well as the E.R. effect for P as a result of soil erosion. Attempts have been made to correlate P sorption with the clay content of soils (Williams et al., 1958). Correlations between P sorption and clay content after removal of Fe and Al oxides and hydrous oxides often have been poor. Better correlations may be expected if P sorption is related to the content of water-dispersed clay. The sorption of P by water-dispersed clay and silt of soils has obvious implications to reactions occurring between dissolved and particulate P in surface runoff and streams.

Sorption of inorganic P by $CaCO_3$ has also been demonstrated (Cole et al., 1953). The nature of the surfaces of calcite in calcareous soils may be very different from those of pure calcite (Buehrer and Williams, 1936; Lahav and Bolt, 1963; Syers et al., 1972).

Consequently, three basic forms of inorganic P may exist in unfertilized soils (Syers and Walker, 1969; Williams and Walker, 1969), namely apatite, which is a discrete phase P compound; P sorbed on the surfaces of Fe, Al and Ca soil components (non-occluded), and P present within the matrices of Fe and Al components (occluded). In fertilized soils, a variety of P fertilizer-soil reaction products may exist as transient phases. As the solubility product of pure apatite in water is low (0.03 µg P/ml at pH 7, Stumm, 1964) and the P held within the matrices of Fe and Al components virtually chemically immobile, except under reducing conditions in the case of Fe, major emphasis should be

directed towards the reactions involving P in solution and that sorbed on the surfaces of Fe, Al and Ca components as well as the release of P due to dissolution of fertilizer-soil reaction products.

Elucidation of the composition of soil organic P is restricted by lack of extractants capable of removing organic P from soils in a relatively unaltered form and by the inadequacy of current methods for mildly degrading extracted organic P-organic matter complexes. Existing data indicate that most of the organic P in soils is associated, in an ill-defined manner, with the humic and fulvic acid complex of soil organic matter (Anderson, 1967). Of the specific forms of organic P that have been identified in soils, inositol phosphates are present in largest relative amounts, comprising up to 60% of the total organic P (Anderson, 1967; Cosgrove, 1967; McKercher, 1969). Other specific organic P compounds are present in soil in much lower quantities: nucleic acids account for 5 to 10%, and other phosphate esters such as phospholipids, sugar phosphates, and phosphoproteins for less than 1 to 2% (McKercher, 1969).

3.5.4.2 Sorption of Dissolved P by Soils

The sorption of dissolved inorganic P by soils may be described by sorption isotherms. Numerous workers have shown that sorption may be described by some of the adsorption isotherms developed to describe gas adsorption by solids (Russell and Prescott, 1916; Olsen and Watanabe, 1957; Rennie and McKercher, 1959; Syers et al., 1973). Similar observations have been made for the sorption of inorganic P by soil components such as kaolinite and short-range-order Fe and Al oxides and hydrous oxides (Gastuche et al., 1963; Muljadi et al., 1966; Kafkafi et al., 1967). Although these studies have been useful in describing relationships between various soils and soil components with respect to their P sorption capacities, they have provided little information regarding P sorption behavior from solutions containing the low dissolved inorganic P concentrations characteristic of most soil-water ecosystems, largely because of the high levels of added P used (Ryden et al.,

(1972b). Furthermore, Syers et al. (1973) obtained two linear Langmuir relationships which intersected at equilibrium P concentrations varying from 1.5 to 3.2 μg P/ml in the equilibrium solution for three contrasting soils; an observation which probably invalidates interpretations of P sorption made from many previous studies were high levels of added P were used.

The study of White and Beckett (1964), conducted at initial dissolved inorganic P concentrations, comparable to those existing in soil-water ecosystems, provides a useful basis for understanding the interactions between aqueous and particulate phases of P in runoff and streams. White and Beckett (1964) defined the intersection of the P sorption isotherm and the abscissa, the "equilibrium phosphate potential" ($1/2\ pCa + pH_2PO_4$), abbreviated to the "equilibrium P concentration" by Taylor and Kunishi (1971). The intersection is equivalent to the inorganic P concentration in the ambient aqueous phase when there is no net sorption or release of P, i.e., $\Delta P=0$. This is a point of reference which provides a predictive estimation of sorption or release of P, should the P concentration in solution change. Furthermore, the average slope of the sorption curve over a given final P concentration range provides information on the ability of the soil to maintain the P concentration of the equilibrium P concentration. The steeper the slope, the closer will the final P concentration be to the equilibrium P concentration. The slope of the curve, although not related to total P sorbed, is related to the extent to which that soil may sorb P over the concentration range considered. The potential of this approach in predicting the chemical mobility of P in soil-water ecosystems is clearly evident and has been used with regard to streams by Taylor and Kunishi (1971) and Ryden et al. (1972a, b) for rural and urban soils, respectively.

The desorption of sorbed P is not as simple as may be inferred from the sorption-release relationships obtained by White and Beckett (1964). In fact very few studies have been reported regarding the desorption of sorbed P. Syers et al. (1970) observed that, for a range

of soils with widely differing P sorption properties, those which sorbed the largest amounts of P desorbed the smallest percentages of that sorbed during a subsequent desorption step. A similar relationship was noted by Ryden et al. (1972a). These studies, however, involved a desorption step following sorption of P from solutions containing P concentration in excess of those commonly found in soil-water ecosystems.

In studies involving the sorption of P by kaolinite from solutions containing realistic inorganic P concentrations. Kafkafi et al. (1967) observed that initially all the sorbed P was isotopically exchangeable. During a subsequent washing or desorption step, however, a portion of the sorbed P became nonexchangeable, or "fixed", this portion being dependent upon the amount of P sorbed, the number of washings and the nature of the previous P sorption cycle. P sorption was represented by either one-step P sorption from a range of solutions of different initial P concentration or by successive additions of small amounts of dissolved inorganic P. Both these forms of P sorption, as well as an effect analogous to washing, could occur in soil-water ecosystems.

Although the mechanisms involved in the retention of organic P by soils have not been established fully, there is evidence that inositol hexaphosphate, and possibly other organic P compounds, are retained by a precipitation rather than by a sorption reaction. Nevertheless, removal of dissolved organic P from solution appears to be a rapid process. Pinck et al. (1941) reported that many commonly occurring water-soluble organic phosphates, e.g. salts of glycerophosphate, hexose diphosphate, and nucleic acids, become non-extractable with water at almost the same rate and as completely as dissolved inorganic P. The retention of water-soluble organic P by sorption reactions may occur by at least two basically different mechanisms (Sommers et al., 1972). Goring and Bartholomew (1950) observed that removal of "free iron oxides" considerably reduced the amount of fructose-1, 6-diphosphate sorbed by subsoil material, suggesting that the sorption of organic P may occur through orthophosphate groups by a similar mechanism to that for inorganic P.

It is possible that organic P can be retained by interaction of the organic moiety of the phosphate ester with inorganic soil components. For example, nucleic acids and nucleotides are protonated at a pH 5 (Jordan, 1955) and could consequently be retained on clay surfaces by displacement of exchangeable cations. Furthermore, physical sorption, also through the organic portion of the molecule, is possible, particularly if the molecular weight of the compound is high, as suggested by Greenland (1965). In such cases retention is weak and is accomplished by van der Waals and ion-dipole forces. Greaves and Wilson (1969) have implicated physical adsorption in the retention of nucleic acids by montmorillonite. It is also possible that retention occurs indirectly through other soil organic compounds such as fulvic and humic acids after interaction of organic phosphates with these species (Martin, 1964).

The desorption of sorbed organic P has not been extensively studied. The hypothesis that inorganic P added to soils displaces sorbed organic P to solution (Latterell et al., 1971) was not supported by the data presented by Wier and Black (1968). Although organic P may be leached from soils, it appears that a large proportion of that removed may not be in a dissolved form. After incubating sucrose with ammonium nitrate in the upper portion of a calcareous soil, Hanapel et al. (1964) found that most of the organic P removed by leaching was present in a particulate rather than a dissolved form.

3.5.4.3 Chemical Aspects of P in Subsurface and Ground-Water Runoff

Losses of P in subsurface and ground-water runoff have been considered minimal in the past, but as discussed in section 3.4 such losses can amount to a significant proportion of losses from agricultural land, and possibly a major proportion from forest lands. The supposition that P losses in subsurface and ground-water runoff are low probably stems from the concept of P immobility based on the P sorption properties of soils using added inorganic P concentrations far in excess of those normally present in the soil solution.

It is of interest to note that many of the reported mean concentrations of dissolved inorganic P in subsurface runoff are within the range of values expected to be maintained in the soil solution. Pierre and Parker (1927) reported values ranging from 0.020 to 0.350 µg P/ml, with an average of 0.090 µg/ml, for several surface soils from the southern and midwestern states of the USA. These workers also noted that dissolved inorganic P concentrations could be maintained at a fairly constant level. Barber et al. (1962) reported similar values for the upper 15 cm (6 inches) of 87 soils from the midwestern USA, with an average of 0.180 µg P/ml; the frequency distribution of the values obtained, however, suggested a mode of between 0.040 and 0.060 µg P/ml.

As water percolates through the soil profile, there tends to be a "chemical sieving" of dissolved inorganic P (Black, 1970). This arises as a result of the sorption of inorganic P by soil components. The low concentrations of P found in ground-water runoff, which has experienced the maximum effects of deep percolation with concomitant increase of contact with P-deficient particulates of the subsoil, are undoubtedly a direct result of the chemical sieving effect. This is illustrated by other data presented by Barber et al. (1962). For the same 87 soils mentioned previously, the average dissolved inorganic P concentration at a depth of 46-61 cm (18 - 24 inches) was 0.089 µg/ml, less than half that for the upper 0-15 cm (0 - 6 inches). A similar effect is observed in results presented by Ryden et al. (1972a) for the P sorption properties of successive soil horizons of Miami silt loam.

The concentration of dissolved inorganic P in subsurface and ground-water runoff will depend on the nature and amounts of P-retaining components in the profile, the surface area exposed to percolating waters, and the ease of percolation which affects the contact time of dissolved inorganic P with the retaining components. In studies of P leaching through columns of organic soils, in the laboratory, Larsen et al. (1958) observed that P retention, measured by ^{32}P autoradiographs, was closely correlated with the total hydrous Fe and Al oxide ("sesquioxide") content. Similarly. losses of P due to leaching through a deep silicious sandy soil were

demonstrated in W. Australia by Ozanne (1962). When 225 kg/ha (209 lb/ac) of ^{32}P-labelled superphosphate was broadcast during winter on a fallow sandy soil, over 50% of the P had penetrated to more than 1 m (3 ft) below the surface within 38 days, during which 230 mm (9 inches) rain had fallen. Ozanne (1962) also demonstrated that the potentially large losses of P to subsurface and ground-water runoff from sandy soil compared to that from loamy soils were due to quantitative rather than qualitative differences in P-retaining components.

Although major emphasis has been placed on P losses in surface runoff, it appears that losses of P to subsurface and ground-water runoff, although of little significance from an agricultural standpoint, may under certain conditions constitute a significant loss of P from agricultural watersheds in terms of the P enrichment of surface waters. Losses of P to subsurface and ground-water runoff are even more difficult to evaluate than those in surface runoff and demand further investigative attention.

3.5.4.4 Chemical Aspects of P in Streams and Stream Bed Sediments

As discussed previously, surface runoff from agricultural land constitutes a heterogeneous and relatively short-lived system. Any attempt to consider the distribution and chemical mobility of P between solid and aqueous phases before entry into the receiving stream would be pointless as a new and more homogeneous system is rapidly established. Surface runoff in urban areas is somewhat different because in most cases it is channellized shortly after origin by alteration of surface drainage patterns, and under such circumstances it is analogous to a stream in an artificial channel. Consequently, the chemical mobility of P will be discussed from the standpoint of the stream environment.

The potential of suspended particulates derived from eroding soil to modify the dissolved inorganic P concentration of streams has been suggested by Taylor (1967) and Biggar and Corey (1969). Wang and Brabec (1969) also implied that inorganic P was sorbed by suspended

particulate material from observations of dissolved inorganic P concentrations in the Illinois River at Peoria Lake.

An evaluation of the possible effects of eroded soil materials on the dissolved inorganic P concentrations of streams may be obtained from P sorption studies (Taylor and Kunishi, 1971; Ryden *et al.*, 1972a, b). It is essential, however, that conditions realistic of those existing in streams are used if meaningful results are to be obtained (Ryden *et al.*, 1972a). Widely differing interpretations can be made as solution:soil ratios and initial P concentrations are changed from those conventionally used in P sorption studies to those realistic in terms of the stream environment. In an investigation of P sorption in a Miami silt loam soil by Ryden *et al.* (1972), the data suggested that inorganic P released from the A1 horizon, which contained a P fertilizer-soil reaction product, would be largely resorbed by the noncalcareous B1 horizon and to some extent by the calcareous 3C1 horizon, should the horizons erode together. Sorption studies employing low initial added inorganic P concentrations and a wide (400:1) solution:soil ratio indicated that the B1 horizon has a much lower ability to remove dissolved inorganic P from solution than expected, this being equal to or only slightly greater than that of the 3C1 horizons. It was found (Ryden *et al.*, 1972b) that the A1 and 3C1 mixtures were able to maintain lower dissolved inorganic P concentrations than the A1 and B1 mixtures. The conditions used by Ryden *et al.* (1972a, b) to predict the potential of eroding soils to modify the dissolved inorganic P concentrations of streams gave results which were comparable to those obtained in simulated stream systems using a solution:soil ratio of 1000:1. This is equivalent to a sediment concentration of 1000 mg/l, which lies well within the range of values cited by Guy and Ferguson (1970) and Johnson and Moldenhauer (1970).

The P sorption studies reported by Taylor and Kunishi (1971) and Ryden *et al.* (1972a, b) involved closed systems, i.e., soil in contact with the same aqueous phase. This may be justified on the grounds that the wash load of a stream travels at the same velocity as the water in which it is suspended (Johnson and Moldenhauer, 1970) as discussed previously.

Sorption studies may be used to provide reasonable estimates of dissolved inorganic P concentrations in streams, under various flow conditions, draining rural watersheds. Taylor and Kunishi (1971) observed that dissolved inorganic P concentrations, during base flow of a stream draining a small agricultural watershed in Pennsylvania, were in the range of 0.040 to 0.060 µg P/ml, values which were close to those predicted from P sorption studies using stream bank sediment and subsoil material. During periods of surface runoff, predicted dissolved inorganic P concentrations would be in excess of 0.20 µg P/ml for the surface soil used by Taylor and Kunishi (1971) and 0.10 µg P/ml for that used by Ryden et al. (1972a) due to release of P from eroded surface soil; however, predictions from the work of Taylor and Kunishi (1971) are based on the use of a narrow (10:1) solution:soil ratio. The ability of eroding stream bank material or resuspended stream bed sediment to resorb inorganic P released to solution should not be ignored (Taylor and Kunishi, 1971).

It is important to distinguish between the quantities of various types of soil materials expected to enter streams in urban as opposed to agricultural surface runoff. In agricultural areas, surface runoff will carry primarily surface soils to receiving streams. Surface soils may frequently contain fertilizer-soil reaction products capable of producing significant increases in dissolved inorganic P concentrations, due to their dissolution (Ryden et al., 1972a). In urban areas, however, land under development, which is prone to severe erosion, is frequently graded, exposing some or all horizons of the area profile to potential erosion. Dissolved inorganic P concentrations of receiving streams in urban areas may be sufficiently high that the addition of eroded soil material may cause a reduction in the dissolved inorganic P concentration.

An approach similar to that used by Taylor and Kunishi (1971) and Ryden et al. (1972a, b) could be used to identify other diffuse sources of potential P enrichment within a watershed. The approach could be particularly useful for estimating the potential of various forms of urban detrital material to influence the dissolved inorganic P concentrations of surface runoff.

One diffuse source of considerable importance is the leachate from leaves, particularly during the autumn. An appreciable percentage of the total P in leaf tissue may be in a water-soluble form. Ash leaves may contain 61.5% of total P as water-soluble inorgaic P (Nykvist, 1959). Cowen and Lee (1972) observed that 44 and 120 µg soluble inorganic P per g air-dry weight of fallen oak and poplar leaves, respectively, could be leached by 1 liter of distilled water percolating at a rate of 8.4 ml/min. Greater amounts of P were released from oak leaves during consecutive leaching cycles, and after fragmentation of whole leaves. Similar experiments were conducted by Timmons et al. (1970) using agricultural crop residues. These were leached in a fresh condition and after dyring, and freezing and thawing cycles. The data suggest that the leaching of crop residues is most likely to contribute to the dissolved inorganic P concentration of streams during spring thaw when, after numerous freezing and thawing cycles, the residues will be carried over frozen ground in surface runoff. When greater infiltration can occur, a portion of the leached P may be retained in the soil due to sorption.

Little is known of the chemistry of stream-bed sediment although it is conceivable that it is similar to that of the subsoil of the surrounding area (Taylor and Kunishi, 1971). Consequently, P sorption studies using subsoil material may provide some information on the role of stream-bed sediment in regulating the dissolved inorganic P concentration due to its suspension during turbulence. This would be particularly true in watersheds, with little contribution to stream-bed sediment as a result of surface runoff. In watersheds where surface runoff is a regular occurrence, however, stream-bed sediment is expected to have a significant contribution from surface horizon soil material, and the latter could contribute to base flow concentration of inorganic P.

Care should be taken, however, in the extension of the P sorption properties of field soils to stream-bed sediment. Hsu (1964) observed that the amount of inorganic P sorbed by soil after storage for 1 year in a continuously wet condition, increased from 69 to 99 µg P/g soil.

The increased sorption was attributed to release of Fe to solution from crystalline phases due to the development of localized reducing conditions during storage, and reprecipitation of "ferric hydroxide" on contact with more aerobic conditions. The redox status of stream-bed sediments does not appear to have been studied, but it is reasonable to suggest that reduction occurs at depth in the sediment with the possibility of crystalline ferric components being transformed to short-range-order ferrous forms. The importance of short-range-order oxides and hydrous oxides of Fe in the sorption of inorganic P has already been discussed. The possible transformation of Fe from crystalline to short-range-order forms represents the first stage of the more aggressive transformations which occur in lake sediments under anaerobic conditions (Shukla et al., 1971).

The observation of Kafkafi et al. (1967) that the washing of kaolinite, on which P had been sorbed, produces a "pool" of nonexchangeable P is also of direct relevance to the dynamics of P in stream-bed sediments, assuming a similar effect occurs. Stream-bed sediment with associated sorbed P could undergo a series of steps equivalent to sequential washing due to resuspension and settling as a result of minor turbulence. The observations of Kafkafi et al. (1967) suggest that sorbed P could become progressively less exchangeable and may constitute an essentially permanent removal of dissolved inorganic P from streams.

When stream-bed sediment contains eroded fertilized soil materials, however, a different situation may prevail. Ryden et al. (1972a) showed that release of P from a surface soil horizon by repeated washing with P-free 0.1M NaCl initially followed first order kinetics, suggesting that release was due to the dissolution of solid phase P, probably a fertilizer-soil reaction product.

3.5.4.5 Forms of P in Runoff and Streams

In many studies concerned with various aspects of P in runoff and streams there has been a tendency to measure total P. The measurement

of total P discharged by streams does not provide any indication of the amounts of P available for aquatic plant growth. Consequently, the forms of P measured in streams that enter a lake or reservoir are of direct importance in assessing the impact of runoff-and stream-derived P on a body of standing water. Dissolved inorganic P is one of the obvious choices because this form of P is directly available for biological utilization. Objections to the measurement of dissolved inorganic P, as is conventionally determined, have been raised by Frink (1971) on the basis that distinction between dissolved and particulate forms is based on filtration through a 0.45 µm filter. Although it is possible that filtration does not strictly differentiate between dissolved and particulate P, it provides a more realistic measure in terms of the effects of runoff-and stream-derived P on the biological productivity of standing waters than the measurement of total P.

Vollenweider (1968) has also pointed out the necessity of distinguishing between total P and dissolved forms of P because it is possible that P exports from some watersheds occur mainly in biologically unavailable forms, such as apatite. He notes that P exports from the Alpine portion of the Rhine Basin amount to 1.45 kg/ha/yr (1.29 lbs/ac/yr). As this is mainly in the form of apatite, however, the contribution of biologically available P to Lake Constance is relatively small. In other regions it appears that a high proportion of particulate inorganic P in streams draining urban and rural watersheds may in fact be apatite. Eroding urban soils in the Lake Mendota watershed, Wisconsin, contain between 6 and 80% of the total inorganic P as apatite, with amounts exceeding 50% in the lower B and C horizons (Ryden and Syers, unpublished data). For the same soil materials, Sagher and Harris (1972) observed that algal cultures suffered P starvation when the sole P source in the growth medium was C horizon material, indicating the very low biological availability of the P present in apatite.

Chemical fractionation schemes have been used to determine the forms of inorganic P in soils. These schemes evolved from the observations of Chang and Jackson (1957) that certain chemical reagents were able to

solubilize inorganic P contained in various synthetic phosphates and phosphate minerals. Recent workers (Bromfield, 1967; Williams et al., 1971a, b; Syers et al., 1972) have questioned the validity of the separation of inorganic P into Al-, Fe-, the Ca-bound forms, as proposed by Chang and Jackson (1957). Providing that the problems inherent in inorganic P fractionation schemes are recognized, useful interpretations may be drawn from the data obtained. The form of particulate inorganic P which is expected to have the greatest potential impact on the biological productivity of standing waters is that which is non-occluded. Part of the non-occluded inorganic P associated with ferric components is released into solution when anaerobic conditions develop subsequent to sedimentation. Appropriate inorganic P fractionation schemes applied to suspended stream sediments may provide a more meaningful measure of the forms and amounts of particulate inorganic P carried in streams. As pointed out by Taylor et al. (1971), suspended sediment concentrations are frequently not high enough to provide adequate amounts of material in a manageable volume of water. Evaluation of the forms of P in soil materials which are known to be transported to streams in surface runoff may overcome this problem to some extent. In the case of eroding soil materials, the inorganic P fractionation schemes should not be applied to the whole soil, due to the E.R. effect resulting from erosion. Water-dispersed particle size separates should be used.

In spite of the possible errors involved in a dissolved-particulate P split based on filter pore size, it seems that in the majority of cases the most meaningful and useful measurements of P in runoff are dissolved forms, particularly dissolved inorganic P. Frequently dissolved forms of P account for a major percentage of total P (Sylvester, 1961; Sullivan and Hullinger, 1969), whereas dissolved inorganic P in many cases constitutes a major proportion of the total dissolved P. It should be noted that dramatic changes can occur in the concentration of dissolved inorganic P and other P fractions after sample collection, even after only a short period of time. In some cases when samples are not analyzed immediately after collection, the only valid P parameter that can be measured is total P.

3.5.5 Chemical Factors Affecting the Mobility of N

3.5.5.1 Nature of Soil N

Most of the N in soils (perhaps more than 95% of the total N) is bound in organic forms (Biggar and Corey, 1969; Bartholomew and Clark; 1956; Keeney, 1970). The organic N consists of the biologically important nitrogenous compounds (e.g. proteins) and N contained in humic matter. Amino acid N (including proteins) accounts for 20 to 50% of the organic N. About 4 to 10% of the organic N occurs in amino sugars. A wide range of other naturally occurring compounds have been found, but these compounds account for only about 1 to 2% of the total organic N. Consequently, about one-half of the organic N can not be accounted for in known biological compounds. Apparently, during chemical and microbial transformations of soil organic matter, microbially synthesized products are converted to more stable humic forms, including humic and fulvic acids.

The main forms of inorganic N in soils are NH_4^+ and NO_3^-, both of which are available to plants. Small amounts of NO_2^- are sometimes present. The NO_3^- ion shows little tendency for sorption onto soil particles while NH_4^+ is held as an exchangeable ion on soil cation exchange sites. Soils also retain some NH_4^+ within clay lattices. This clay-fixed NH_4^+ is relatively unavailable to plants and microorganisms. The amounts of NH_4-N and NO_3-N present in a soil at a given time generally represent less than 0.1% of the N in the soil. However, organic N is converted slowly to NH_4-N and subsequently to NO_3-N through microbial processes.

3.5.5.2 Transformations and Mobility of Soil N[1]

The predominant storage reservoir of N in soils is the organic matter. Added plant and animal debris are attacked readily by heterotrophic microorganisms with some of the N being added to the organic matter

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[1] Based on the review of Keeney (1970).

reserve, and some being liberated as inorganic (predominantly NH_4^+) waste products. Depending on the supply of available energy (carbon), inorganic N can be immobilized in microbial tissue which continually adds to the supply of organic matter. Concurrently, soil organic matter is continually depleted through microbial breakdown. The dynamic nature of nitrogen turnover in soils does not at first appear to be consistent with the relative stability of soil organic nitrogen. In soils, only about 1 to 3% of the organic N is available for plant uptake in a growing season (Bremner, 1967). The stability of organic N in soils is generally attributed to formation of herterocyclic N compounds, clay-organic matter complexes which are resistant to microbial attack, and to lack of sufficient carbonaceous energy material for complete breakdown (Bremner, 1967).

The processes leading to formation of NO_3^- by microbial action in soils and waters can be grouped conveniently into ammonification and nitrificiation reactions (Delwiche, 1970). Ammonification (organic N to NH_4^+) is conducted by numerous heterotrophs under a wide diversity of environmental (pH, temperature, moisture) conditions (Bartholomew, 1965; Campbell and Lees, 1967). With few exceptions, nitrification (NH_4^+ to NO_2^- to NO_3^-) is carried out by strict chemoautotrophs. Although several heterotrophic nitrifiers have been identified, their importance in the nitrification process is believed to be minimal except in organic-rich environments such as composts and manure piles. Because of its agronomic importance, the nitrification process has been studied widely and reviewed extensively (Quastel and Scholefield, 1951; Campbell and Lees, 1967; Keeney and Gardner, 1970). In contrast to ammonification, nitrification is easily inhibited by unfavorable conditions. Optimal nitrification occurs at pH values near 7 and in the presence of adequate oxygen. The nitrification rate increases with temperature to 35 C, but is inhibited completely at greater than 40 C. Formation of NO_2^- from NH_4^+ (Nitrosomonas spp.) can be inhibited by numerous organic and inorganic compounds. Oxidation of NO_2^- to NO_3^- (Nitrobacter spp.) can be inhibited by the high pH-free NH_4^+ conditions occurring in the

area around urea or anhydrous ammonia fertilizer bands giving rise to phytotoxic NO_2^- accumulation in some alkaline soils following N fertilization.

The salts of NO_3^- are water soluble and nitrate does not react to any great extent chemically or physically with soil constituents to form insoluble products. As a result, NO_3^- in soils is readily available for plant uptake or loss to ground waters by leaching. It is this unique property of NO_3^-, coupled with its toxicological and nutrient properties, that make it of such concern.

The process of denitrification (reduction of NO_3^- or NO_2^- to elemental N and N oxides) provides the major pathway whereby N is cycled back into the atmosphere. Denitrification can occur by two distinct pathways. Biological denitrification occurs when oxygen tensions reach very low levels (through flooding of soils and utilization of oxygen by decomposition of organic matter) and the heterotrophic microorganisms must utilize other electron acceptors. With few exceptions, carbonaceous material must be present to provide sufficient energy (Broadbent and Clark, 1965). In acid soils and waters, NO_2^- is extremely unstable and decomposes to yield NO_3^- and gaseous N oxides, or reacts with soil constituents with the fixation of N in organic matter and formation of N oxides (Bremner and Nelson, 1968; Brodbent and Clark, 1965). These latter reactions have been termed "chemo denitrification" as they do not involve microbial transformations directly and can occur in arable soils. The importance of denitrification reactions in soils and waters cannot be overstated, and these complex and poorly understood reactions must be evaluated to enable recommendations to be made on methods of biological NO_3^- removal from soils and waters as well as N fertilizer rates.

A point worthy of emphasis in the N cycle is that once N enters the soils from any source, its identity is lost because of the complex series of transformations it may undergo. Thus, investigations which attempt to pinpoint sources of NO_3^- polluting water supplies are particularly difficult and, unless properly conducted, subject to many errors (Keeney, 1970).

3.5.5.3 N in Surface Runoff, Streams, and Stream Bed Sediments

Surface runoff occurs whenever the rate of precipitation exceeds the rate of infiltration (Keeney and Walsh, 1972). Soil texture, slope, moisture content, physical character of the surface, soil and water conservation practices, and vegetative cover affect the infiltration rate. Infiltration is usually quite rapid with coarse-textured sandy soils and with well-aggregated, fine-textured soils having large pores and no impervious subsoil layers. The energy associated with the impact of falling raindrops is an important factor in the breakdown of soil aggregates, increasing runoff. Any cover, vegetative or mulch, that absorbs raindrop impact tends to protect the soil structure at the surface and thus promote infiltration.

The high water solubility of NO_3^- and NH_4^+ tends to favor movement of these ions into the soil before runoff occurs (Bailey, 1968; Armstrong and Rohlich, 1970; Biggar and Corey, 1969). Under some conditions, the concentrations of NO_3-N and NH_4-N in runoff may be lower than that in rain water. However, Timmons et al. (1970) noted that leaching of soluble nutrients from cover crops could contribute substantial concentrations of N and P to surface runoff. Furthermore, data on N concentrations in streams receiving runoff from agricultural areas shows that appreciable amounts of N may be transported in surface runoff (See Section III-4). For example, Jaworski et al. (1969) observed NO_3-N concentrations as high as 3 mg/l and frequently near 1 mg/l for Catoctin Creek, Md., a stream draining a predominantly agricultural area. In basins containing large areas of agricultural lands. The nutrient contributions to the receiving water may be appreciable even though the amount per unit of land area (kg/ha) is small.

Dissolved and particulate N compounds in runoff are potential sources of N to streams and lakes. Due to rapid nitrification in aerobic systems, concentrations of NH_4-N in streams are generally low compared to NO_3-N. Amounts of dissolved and particulate organic N are variable. Total N (including particulate) is frequently considerably higher than NO_3-N.

The N that reaches surface waters may be immobilized in the sediments and thus only partially available (Keeney, 1972). Conversly, some of the N may be released from the sediments by microbial mineralization. Soil erosion is a selective process, and eroded material contains 2.7 to 5 times the total N content of the original soil (Hensler and Attoe, 1970).

The effect of fertilizer N for continuous corn on runoff and nitrogen losses has been studied in Missouri. It was found that 0.5 m^3/ha (7 ft^3/ac) of runoff resulted from 11.4 cm (4.5 inches) of rain and that a runoff loss of only 0.01 kg/ha (0.01 lb/ac) of NO_3-N resulted from application of 198 kg/ha (177 lb/ac) of fertilizer N (Smith, 1967).

Minnesota workers found soil losses of 7.7 metric tons/ha (7 tons/ac) for fallow conditions, 2.0 (1.8) for continuous corn, and less than 0.55 (0.5) for crop rotations. The N in this sediment amounts to 63.5 (56.7), 12.9 (11.5) and less than 5.6 kg/ha (5 lb/ac), respectively. The dissolved N in the runoff water ranged from 0.75 to 1.21 kg/ha (0.67 to 1.08 lb/ac) for the continuous corn and for corn and oats in the rotation. However, the amount of dissolved N in the runoff water from hay was 3.47 kg/ha (3.10 lb/ac). This was probably due to the leaching of N from the frozen hay residues (Timmons et al., 1970). Wisconsin researchers have also noted that the content of soluble N in runoff water from sod plots was about 5 times greater than that from fallow plots, while work on rotational land with slopes ranging from 3 to 20% has shown that runoff losses of NO_3-N were less than 1 kg/ha (0.9 lb/ac) (Hensler and Attoe, 1970).

3.5.5.4 Nitrogen in Subsurface and Ground-Water Runoff

Water that infiltrates into the soil ultimately is either evaporated, transpired, seeps along impervious layers, or percolates to the water table. The amount of leaching at a particular location is largely related to conditions affecting evapotranspiration, soil physical conditions, and to precipitation distribution and intensity. As reviewed by Keeney (1970), the negatively charged nitrate ion (NO_3^-)

is soluble in water nad generally moves with the water. The positively charged ammonium ion (NH_4^+), however, is held in exchangeable form by clay minerals and soil organic matter which greatly restrict its movement. Sediments and organic matter do not readily move downward in the soil; bacteria and pesticides are generally adsorbed by soil particles; and the bulk of most plant nutrients are held in exchangeable, non-exchangeable, adsorbed or organic forms that are resistant to leaching. Thus, NO_3-N is usually of greater concern in groundwater pollution.

Nitrate-N tends to accumulate at the soil surface during prolonged droughts due to evapoaration. Rain following a drought moves NO_3-N into the root zone, or beyond in the case of excessive rains. Incorporation of carbonaceous crop residues tends to retard production of NO_3-N and movement due to microbial immobilization of N.

In Colorado, the average kg/ha (lb/ac) of NO_3-N found in the soil to a depth of 6.1 m (20 ft) for various cropping practices were: alfalfa, 88 (79); native grassland, 100 (89); cultivated dry land, 292 (261); irrigated fields not in alfalfa, 569 (508); and corrals, 1,608 (1,436) (Stewart et al., 1968). Variations in the NO_3-N content of the corral soils were directly related to teh degree of aeration. For the irrigated fields the annual losses of NO_3-N to the groundwater were 28 to 41 kg/ha (25 to 37 lb/ac). Stout and Burau (1967) found that soil fertility levels are the prime factors determing the amount of NO_3 below the root zone, showing that NO_3 is largely from natural sources.

In preliminary study in Wisconsin (Olsen, 1969), undisturbed, cultivated, and poorly drained and well drained barnyard soil profiles were analyzed for their NO_3-N content. The respective profiles averaged 55 (49), 231 (206), 165 (147), and 134 kg/ha (120 lb/ac) of NO_3-N to a depth of 2.4 m (8 ft). The content tended to decrease with soil depth, apparently due to plant uptake, denitrification, and microbial immobilization processes.

On irrigated sands, loss of NO_3-N is closely related to the amount of fertilizer N and irrigation water applied (Olsen, 1969).

In Wisconsin 30.5 cm (12 inches) of rain or irrigation water during a 5-week period following application of ammonium nitrate to a fallow Plainfield sand caused most of the NO_3-N to move down to the 1 to 1.5 m (40 to 60 inch) depth. An additional 15 cm (6 inches) of precipitation during the following month moved most of the NO_3-N below the 1.8 m (6 ft) depth. In a related study, the annual application of 336 kg/ha (300 lb/acre) of N as ammonium nitrate to a silt loam soil for corn resulted in the accumulation of excessive amounts of NO_3-N in the subsoil, compared to very little accumulation from the 112 kg/ha (100 lb/acre) rate. The average annual precipitation was about 81.3 cm (32 inches) and the rate of movement of NO_3-N was about 41 cm (16 inches) annually. These data suggest that pollution of the groundwater with NO_3-N can be greatly reduced by limiting the rate of fertilizer N to approximate the needs of the crop.

Hedlin (1971) found that high NO_3-N concentrations in groundwater were most likely where repeated heavy applications of organic matter or nitrogenous materials take place (e.g. in the vicinity of farmsteads, towns or villages). He also noted that the NO_3-N concentration in the groundwater decreases rapidly as one moves away from the source, indicating either dilution by NO_3-free water or denitrification or both. Little is known of the persistence of NO_3 in groundwater and evaluation of the possibility of denitrification in groundwaters would seem to be a needed area of investigation.

3.6 Natural Renovation Mechanisms Available to Remove Nutrients From Agricultural Runoff

The physical and chemical processes controlling the mobility of soil phosphorus and nitrogen are reviewed in Section 3.5. Included in this section is a discussion of natural mechanisms which tend to limit the movement of P and N in agricultural runoff. These processes include the soil stabilization and sediment deposition, sorption of phosphate by soil particles, retention of NH_4-N by soil cation exchange sites, immobilization of nutrients by plant assimilation, and denitrification of NO_3-N.

Soil erosion is a major factor in the transport of nutrients in agricultural runoff. Thus, mechanisms and practices which minimize erosion and transport of soil particles reduce nutrient transport in runoff. The natural processes of major importance is maintenance of a crop or mulch cover on the soil (Amemiya, 1970).

Phosphate characteristically exhibits a low water solubility in soil systems as a result of sorption and/or precipitation processes (Taylor, 1967). However, the retention of phosphate is a function of soil type and is generally high for fine-textured soils and low for sands. Thus, the success in using sorption by the soil as a mechanism for removing P from runoff will depend on the characteristics of the soil. In well drained silt loam soils, sorption should reduce P concentrations to sufficiently low levels in the absence of excessive loadings of P to the soil.

Nitrification greatly increases the mobility of NH_4-N and organic N in soils and the tendency for N to percolate to ground waters. Conditions for nitrification are generally favorable for agricultural crops. Mobility is best controlled by balancing NO_3-N from the soil but requires anaerobic conditions which are generally unacceptable for crop production.

Practices based on the factors and mechanisms controlling the mobility of nutrients in soils and used to minimize nutrient transport in runoff are discussed in Section 5.

3.7 Probable Changes in Land Use Activities (1980, 2000, 2020)

Changes in the total cropping area for the U.S. region of the Great Lakes Basin were estimated in the Great Lakes Basin Framework Study, Appendix 13, Land Use and Management (1971) and are shown in Table 11. A gradual decrease in cropping area was predicted, amounting to 10 per cent of the present cropping area by 2020. Some changes in areas for crop groups were predicted. Areas in idle cropland, pastures and meadows, and close grown crops were projected to decrease, while row crop area was expected to increase. These projections indicate that there is little land available for conversion to agricultural use in the Basin.

Table 11. Total Cropping Area, Current and Projections for 1980, 2000 and 2020 for Great Lakes Basin[1]

Crops	Total Area							
	1966		1980		2000		2020	
	1000 hectares				(1000 acres)			
Row Crops	3,949	(9,750)	3,849	(9,503)	4,084	(10,083)	4,821	(11,903)
Close Grown Crops	1,458	(3,600)	1,617	(3,993)	1,467	(3,621)	1,284	(3,170)
Pastures and Meadows	2,719	(6,715)	1,995	(4,926)	1,721	(4,239)	1,654	(4,083)
Orchards and Vineyards	243	(600)	161	(398)	165	(407)	185	(456)
Idle Cropland	3,219	(7,947)	3,620	(8,939)	3,319	(8,194)	2,398	(5,920)
Total	11,588	(28,612)	11,242	(27,759)	10,750	(26,544)	10,342	(25,532)

[1]Great Lakes Basin Framework Study, Appendix 13, 1971.

3.8 Projected Seriousness of Future Pollution Problems Derived From Nutrients From Agricultural Lands.

The amounts of nutrients from agricultural lands expected to reach the Great Lakes can be projected from land use predictions (Section 3.7) and estimated loadings per unit area of agricultural land (Section 3.4). These estimates assume that the rate of loading for a given type of crop will not change. Further, as discussed in Section 3.4, the estimated loadings are only approximations and not sufficiently accurate to justify quantitative estimates of changes in loadings due to changes in fertilizer use or other farming practices. Because land use changes are expected to be small, the corresponding changes in nutrient loadings from agricultural lands are also small (Table 12). On this basis, it appears that the nutrient loadings from agricultural land have reached a maximum value and will remain the same, given current agricultural practices. Relatedly, if agricultural technology can be developed to reduce nutrient loadings to surface waters per unit land area, corresponding decreases in nutrient loadings to the Great Lakes can be expected.

While the nutrient loadings from agricultural lands may remain relatively constant, the proportion of phosphorus entering the Great Lakes from agricultural lands as compared to other sources will likely increase substantially. Treatment of municipal and industrial wastes to remove phosphorus is receiving major emphasis. These sources presently account for a high proportion of the phosphorus entering the Great Lakes. Phosphorus budgets for the total Great Lakes system (Great Lakes Water Quality Board, 1973) do not provide estimates of the proportion of the total P loading derived from municipal wastes. Further, discrepancies exist between the budgets presented in the section on water quality and in the section on eutrophication, which includes estimates in reduction in P loading for the period 1971-1973. However, it appears that an overall reduction of about 25% in the P loading to the Great Lakes from the U.S. and Canada was projected by 1973. On this basis, if agriculture previously contributed about 20%, this proportion increased to about 25% if projected reduction was achieved. From another viewpoint, at least an 80 percent

Table 12. Projected Future Loadings of Soluble Nutrients to the Great Lakes from Agricultural Lands.

Crops	Estimated Nutrient Loadings to the Great Lakes							
	1966[1]	1980[2]	2000[2]	2020[2]	1966[1]	1980[2]	2000[2]	2020[2]
	metric tons/yr (tons/yr)							
	Inorganic Phosphorus				Inorganic Nitrogen			
Row Crops	829 (914)	808 (889)	858 (944)	1,010 (1,111)	6,160 (6,780)	6,000 (6,600)	6,370 (7,010)	7,520 (8,270)
Close-Grown Crops	190 (209)	210 (231)	191 (210)	167 (184)	2,400 (2,680)	2,700 (2,970)	2,450 (2,700)	2,140 (2,350)
Pastures and Meadows	598 (659)	439 (483)	379 (419)	364 (400)	6,250 (6,880)	4,590 (5,050)	3,960 (4,360)	3,800 (4,180)
Orchards and Vineyards	51 (56)	34 (37)	35 (39)	39 (43)	379 (417)	251 (276)	257 (283)	289 (318)
Idle Crop Land	161 (178)	181 (199)	166 (183)	120 (132)	12,600 (13,800)	14,100 (15,500)	12,900 (14,200)	9,350 (12,300)
Total	1,829 (2,017)	1,672 (1,839)	1,629 (1,793)	1,700 (1,870)	27,829 (30,559)	27,621 (30,396)	25,437 (28,553)	23,099 (27,418)
Total[3]	4,170 (4,590)	4,050 (4,460)	3,870 (4,260)	3,720 (4,090)	56,800 (62,500)	55,100 (60,600)	52,700 (58,000)	50,700 (55,800)

[1]Values from Table 7 & 8. [2]Based on projected area (Table 11) and current estimated loadings (Table 7 & 8). [3]Based on stream data(Total P and Total N).

loading to Lake Michigan is projected. Point sources were estimated to contribute about 77 percent of the loading prior to treatment (Table 4). If the agricultural contribution was about 20 percent, as estimated for the total Great Lakes Basin, this proportion would increase to about 53% when the projected 80 percent from point sources is achieved. While these values are approximations at best, it is clear that the proportion of the phosphorus loading to the Great Lakes from agricultural lands will increase as treatment is implemented for point sources. This will focus the attention on agricultural sources of phosphorus to the Great Lakes.

Because major changes in the amounts of agricultural land in the Great Lakes Basin are not anticipated, the seriousness of the problem of nutrient loadings to the Great Lakes from agricultural lands in the future, as compared to the present time, will be a function mainly of changes in practices which affect the nutrient loadings per unit area of agricultural land. In this regard, the extent to which nutrient levels in runoff are affected by fertilizer use and other agricultural practices becomes an important question.

Agricultural soils contain varying amounts of native nutrients and nutrients added as fertilizers. The large amounts of fertilizer N and P used, 260,000 and 152,000 metric tons (286,600 and 167,500 tons) respectively, for the Great Lakes Basin, have raised concern over the impact of fertilizer nutrients on loadings to the Great Lakes. Most agricultural experts (e.g. Taylor, 1967; Stanford et al., 1970; Viets 1970, 1971; Nelson, 1972) agree that insufficient information is available from sufficiently controlled situations to accurately assess the contribution of fertilizers to the nutrient loadings to lakes. Further, it is agreed that losses of fertilizer nutrients can be affected by fertilizer use practices.

The amounts of fertilizer nutrients added to the soil are small on an annual basis compared to native nutrient levels. However, over years of continuous fertilizer use, the amounts accumulated can be significant. Because of the low water solubilities of P in soils,

P is transported from soils largely in particulate form by soil erosion. Although soil NO_3-N is water soluble, soil N is largely particulate. Thus, P and N losses are closely related to the P and N contents of the soil and may be increased by fertilization of the soil. A loss of 11.2 metric tons/ha (5 tons/ac) of soil containing 1000 ppm P amounts to a loss of 11.2 kg/ha (10 lb/ac) of P. Furthermore, eroded soil is generally enriched in nutrients compared to the source soil, due to the tendency for nutrients to associate with fine silt and clay fractions. However, prediction of the impact of soil erosion on nutrient loadings to lakes, complicated by the partial deposition of sediment prior to reaching the lake and a lack of information on the availability in lakes of nutrients associated with the particulate phase (Taylor, 1967; Syers et al., 1973).

Although P and N are lost from soil mainly in eroded soil particles, the losses of these nutrients in the soluble fraction is also important. The soluble inorganic forms are directly available to aquatic plants and are more likely to reach the lake than are particulate forms. Losses of nutrients in solution are related in part to the concentrations maintained in solution by soil particles.

Fertilization is expected to increase soluble P concentrations based on soil P sorption isotherms and on the solubility of fertilizers and their reaction products in soil (Black, 1970; Ryden et al., 1973). Generally, fertilizer P reacts in soil initially to form products of relatively high water solubility (Black, 1970). As P diffuses from the fertilizer band, the concentration is decreased as P is sorbed by soil particles. As the amount of P sorbed by the soil is increased, a higher solution concentration is maintained. Consequently, both the formation of fertilizer reaction products and the increase in sorbed P would increase the tendency for loss of P in solution in runoff waters.

While some data have been obtained to show that fertilization can increase soluble P concentrations, the importance of this to P loadings to lakes has not been established. Data cited by Taylor (1967) showed that P concentrations of 13, 34 and 88 µg/l, were obtained when dilute

salt solutions were equilibrated with a silt loam soil fertilized at rates of 0, 244 and 468 kg/ha (0, 218 and 418 lb/ac), respectively. Bolton et al. (1970) obtained P concentrations of 13 and 24 µg/l in tile drainage water from non-fertilized and fertilized corn plots in a 4-year rotation with oats-alfalfa. Concentrations of 26 and 29 µg/l were obtained for non-fertilized and fertilized continuous corn plots. Nelson and Romkens (1970) observed that the P concentration in runoff water was directly related to the rate of fertilizer addition for plots receiving 0, 56 and 112 kg/ha (0, 50 and 100 lb/ac) of fertilizer P and subjected to simulated rainfall. Concentrations ranged from about 50 µg/l for the unfertilized plot to 400 for the highest level of added P. The amount of solution P lost per unit area was also directly related to fertilizer addition rate. It should be noted that the "rainfall" intensity (6.4 cm/hr or 2.5 inches/hr) and duration (1 hr) were high, contributing to a high rate of runoff. Consequently, the losses were higher than normally expected. Holt et al. (1970) reported P concentrations of 8, 9, 16 and 30 µg/l in runoff from plots receiving no fertilizer, fertilizer broadcast and plowed, fertilizer broadcast and disked, and fertilizer broadcast (rates not given), respectively.

The predominant form of N transported in solution is NO_3^-. In contrast to P, NO_3^- is not retained by the soil and moves with runoff and percolating waters. The extent of soluble N transport from the soil is a function of the rates of water percolation, nitrification, denitrification, plant uptake, and immobilization by other processes (see Section 3.4). While instances of enrichment of subsurface waters by fertilizer N are known, the importance of fertilizer N to the loadings of N from agricultural lands to lakes is not well established (Stanford et al., 1970; Viets, 1971; Keeney and Walsh, 1972).

Available information indicates that nutrients in solution contribute significantly to estimated nutrients loadings to lakes from agricultural lands (Section 3.4). Fertilization can increase concentrations of nutrients in solution and thereby increase the amounts of nutrients transported from soil in runoff and percolating waters. On this basis, future problems of nutrient additions to lakes from agricultural lands may be

reduced if emphasis is placed on balancing fertilizer application rates to agricultural crop needs and on achieving the minimum nutrient concentrations in the soil solution for maximum crop production. From another standpoint, considerable nutrient transport occurs through soil erosion and fertilization increases the nutrient content of the soil. Consequently, future problems for lakes can be reduced by avoiding excessive fertilizer use and by minimizing soil erosion.

3.9 Institutional Arrangements Relating to Great Lakes Pollution By Nutrients From Agricultural Lands

The laws, policies, and institutional arrangements pertaining to the U.S. Great Lakes Basin are discussed in detail in Appendix 20 of the Great Lakes Basin Framework Study, 1972. Apparently, programs and policies have not been developed to deal directly with pollution by nutrients from agricultural lands. However, certain agency programs and policies developed for other purposes have a bearing on nutrients from agricultural lands. Most closely related are programs concerned with control of erosion and sediment. The Fedearl Water Pollution Control Act Amendments of 1972 focused attention on diffuse sources of pollution, including sediments. The legislation called for an evaluation of the sources and extent of sediment and associated pollution arising from agricultural and urban lands, and of the legal, economic, and other implications of the implementation of erosion control methodology.

3.9.1 Agencies

The EPA, and USDA have programs concerned with agricultural pollution (EPA, 1971). The EPA is the agency with direct statutory responsibility for programs leading to abatement, prevention, and control of all water quality problems. Within the research and development program, projects are included to improve methods to alleviate or abate all agricultural sources of pollution.

Several USDA agencies are concerned with proper land use and resource conservation practices. These practices result in soil and water conservation and may result in pollution abatement, particularly sediment control. The Agricultural Stabilization and Conservation Service (ASCS) authorizes federal cost sharing with farmers for carrying out soil conservation programs and specific pollution abatement practices. Pollution abatement has been concerned mainly with animal wastes. The Agricultural Research Service conducts land and water resource research programs. Some of these relate to erosion control and pollution by agricultural chemicals. The Cooperative State Research Service administers Federal grant programs for agricultural research at state agricultural experiment stations. All experiments in the Great Lakes states are conducting research on nutrient enrichment of waters. The Soil and Water Conservation Service develops and carries out a national soil and water conservation program through soil and water conservation districts and watershed protection, flood prevention and river basin investigation projects. A survey of small watersheds in the Great Lakes St. Lawrence Region concluded that watersheds were feasible for development for water and land conservation and flood control under Public Law 566 (EPA, 1971). Local sponsoring organizations made application for 74 projects. Planning was authorized for 31 and construction for 21 projects.

Within state governments, various groups such as the Departments of Agriculture are involved in programs related to land use. Generally, these programs inform agricultural committees on farming practices, and other matters, and advise the legislative branch on subjects including agricultural pollutants.

3.9.2 Programs and Policies

Floodplain management may indirectly affect nutrient pollution from agricultural lands located in floodplains. However, management is mainly concerned with uses other than agricultural. All Great Lakes Basin states have zoning enabling legislation. Wisconsin is the only state with a compulsory floodplain zoning program (Great Lakes Basin Framework Study, 1972).

Land drainage (i.e., installation of drains, etc.) is controlled under enacted statutes in six of the Basin states (Great Lakes Basin Framework Study, 1972). Drains may be considered as public waters and subject to pollution abatement laws. Consequently, where runoff waters are channeled through drains, a potential exists for control from the standpoint of the impact of nutrient loadings to surface waters.

The states have soil conservation programs which affect pollution by nutrients from agricultural lands. All Basin states, with the possible exception of Indiana, have statutory provisions to abate or safeguard against sediment and erosion damages (Great Lakes Basin Framework Study, 1972). Erosion is an important aspect of nutrient transport from agricultural lands. Soil conservation districts may be formed in all Basin states to effect soil conservation practices, for example, as a mechanism for erosion and sediment control.

Various programs for controlling pollution from land use activities are in the planning or demonstration stages. Allen County, Indiana, Soil and Water Conservation District is working with EPA in a research and demonstration project to control pollution runoff from farmlands (Great Lakes Water Quality Board, 1973). The Minnesota Governor's Conference with participation by EPA is working to develop a model state act for soil erosion and sediment control. Michigan is considering legislation to control soil erosion. Ohio is considering a cooperative plan among state agencies to control nutrients and sediments from agricultural lands.

Under Wisconsin law, land use regulations through adopted ordinances which affect soil and water erosion, flooding, a sedimentation can be formulated by soil and water conservation districts for land use within the district but outside of incorporated villages and cities. All landowners in the district would be included under such ordinances. A model ordinance for Wisconsin soil and water conservation districts adopting land use regulations for sediment control has been developed. Wisconsin has planned a cooperative research and demonstration project to demonstrate the effectiveness of land control measures in improving water quality and to devise the necessary institutional arrangements

for the preparation, acceptance, adoption and implementation of a sediment control ordinance applicable to incorporated and unincorporated areas on a county-wide basis. The project involves cooperation between ten local, state or federal groups. Washington County in southeastern Wisconsin has been selected as the site for the proposed program.

Section 4

SUMMARY OF RELEVANT ONGOING RESEARCH DEMONSTRATION OR MONITORING PROGRAMS

4.1 Ongoing Research[1]

The nutrient contribution by the agricultural land to surface waters is difficult to quantify, because of the diffuse nature of agricultural sources. Also, the problem is complicated by a multitude of factors, both naturally occurring and man-made, influencing nutrient transport in runoff. The more obvious factors include the amount, distribution, and intensity of rainfall; the amount, quality and application of irrigation water; temperature and evapotranspiration; the amount of runoff and percolation; soil erosion and conservation practices; the physical and chemical nature of the soil; topographic, geological, and hydrological features of the earth's surface; the kinds of crops grown and the methods of crop management; and the kinds and amounts of fertilizer used and the timing of application. These factors act separately and in combination to influence the nutrient losses. Interaction among factors greatly complicates evaluation of the problem (Taylor, 1967; Stanford et al., 1970; Nelson, 1970).

Several approaches currently are used to evaluate the problem, namely studies with single or multiple watersheds, lysimeter plots, drainage plots and/or runoff plots. Monitoring of the nutrient content of streams, ponds, and lakes provides valuable data if done carefully and continuously in well selected areas where there is minimal contamination from outside sources of nutrients.

Generally, the supporting agencies are Environmental Protection Agency (Office of Research and Development), Department of the Interior (Office of Water Resources Research), Department of Agricultural (Agricultural Research Service), Department of Agriculture with Cooperative State Research Service, and State Governments.

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[1] See also Section 3.9, Institutional Arrangements.

The projects currently being undertaken are presented below according to subject matter.

4.1.1 Contribution of Nutrients from Agricultural Land to Surface Waters

Projects involving evaluation of the extent of phosphorus and nitrogen enrichment of surface waters by agricultural land are being undertaken by the following investigators: D. K. Cassell and W. C. Dahnke (North Dakota State University); G. Chesters and D. R. Keeney (University of Wisconsin); B. G. Ellis (Michigan State Univesrity) J. J. Hanway, J. M. Bremner and M. A. Tabatabai (Iowa State University); R. F. Harris and D. E. Armstrong (University of Wisconsin); H. G. Heinemann (U.S. Department of Agriculture); C. C. Hortenstine, L. C. Hammond, and R. S. Mansell (University of Florida); J. M. MacGregor and R. S. Adams (University of Minnesota); B. L. McNeal (Washington State University); L. S. Murphy (Kansas State University); D. W. Nelson and M. J. Romkens (Purdue University); R. A. Olson, J. Muir and E. C. Seims (University of Nebraska); B. L. Schmidt and T. J. Logan (Ohio Agricultural Research and Development Center); D. R. Timmons and R. F. Holt (U.S. Department of Agriculture); E. M. White (South Dakota State University); and P. J. Zwerman, D. J. Lathwell and D. R. Bouldin (State University of New York at Ithaca). One investigator, G. O. Schwab (Ohio State University) is studying specifically the movement of nutrients from agricultural land to Lake Erie.

4.1.2 Contribution of Nutrients by Fertilizers to Surface Waters

Current projects on the role of fertilizers in pollution of surface waters include those conducted by: R. R. Bradford (Alabama Agricultural and Mechanical College); L. A. Douglas (Rutgers, the State Univesrity); E. P. Dunigan (Louisiana State University); J. E. Giddens (University of Georgia); J. W. Gilliam (Univesrity of North Carolina);

J. W. Gilliam and S. B. Weed (University of North Carolina); G. F. Griffin and R. W. Wengel (University of Connecticut); J. J. Hanway, R. M. Shibles, and E. J. Dunphy (Iowa State University); L. H. Hileman (University of Arkansas); D. M. Himmelblau and M. Hildebrand (University of Texas); C. C. Hortenstine, D. A. Graetz and D. F. Rothwell (Univesity of Florida); D. E. Kissel and C. W. Richardson (Blackland Conservation Experiment Station); W. Kroontje (Virginia Polytechnical Institute); G. M. Lessman (University of Tennessee); J. F. Lutz (University of North Carolina); T. M. McCalla and G. L. Schuman (University of Nebraska); J. D. Menzies and G. Stanford (U.S. Department of Agriculture); R. A. Olson, A. D. Flowerday, D. Knudsen and G. A. Peterson (University of Nebraska); A. R. Overman (University of Florida); T. C. Peele (Clemson University); C. E. Scarsbrook (Auburn University); G. E. Smith (University of Missouri); A. Swoboda (Texas A & M University System); G. W. Thomas (University of Kentucky); M. R. Till and D. W. Armstrong (South Australian Department of Agriculture); F. G. Viets and S. R. Olsen (U.S. Department of Agriculture); L. F. Welch, J. D. Alexander and T. E. Larson (University of Illinois); C. W. Wendt, A. B. Onken and O. C. Wilke (South Plains Research and Experiment Center); and W. M. Winant (university of Vermont).

4.1.3 Nitrogen and Nitrates in the Environment

Nitrogen and nitrates in teh soil, plants and surface waters are under investigation by: M. M. Alexander and P. L. Minott (State University of New York); S. A. Barber and D. W. Nelson (Purdue University); A. V. Barker and D. N. Maynard (University of Massachusetts); J. M. Bremner and L. G. Bundy (Iowa State University); J. C. Day (Univesrity of Arizona); Y. Kanehiro (University of Hawaii); D. R. Keeney (Univesrity of Wisconsin); F. E. Koehler and N. K. Whittlesey (Washington State Univesrity); W. D. Lembke, J. K. Mitchell and J. Simon (Unviversity of Illinois); G. D. Lewis, D. V. Naylor and D. W. Fitzsimmons (University of Idaho); A. D. McLaren and R. K. Schulz (University of California); R. J. Miller (University of California); D. W. Nelson (Purdue University); N. K. Peterson

(University of New Hampshire); R. V. Rourke and R. F. Stafford (University of Maine); J. R. Sims (Montana State University); E. R. Swanson (University of Illinois); A. R. Swoboda (Texas A & M University System); J. M. Tiedje (Michigan State University); S. J. Toth (Rutgers, The State Univesrity); T. C. Tucker, G. R. Dutt and R. L. Westerman (Univesrity of Arizona); R. A. Young and R. J. Ruf (University of Nevada); V. V. Volk and M. G. Cropsey (Oregon State University); C. W. Wendt (Texas A & M University System); and F. Wiersma (University of Arizona).

Section 5

NATURE AND AVAILABILITY OF TECHNOLOGY TO COPE WITH NUTRIENTS IN AGRICULTURAL RUNOFF

Soil erosion and sediment transport play a major role in the transport of nutrients in agricultural runoff (Taylor, 1967; EPA, 1971; Sections 3.4 and 3.5). Sediment transport has long been recognized as a problem in agricultural areas, and soil conservation and sediment control practices have received considerable attention (Glymph and Storey, 1967; Amemiya, 1970; EPA, 1971). Generally, technology to control sediment transport is available, but further effort is needed in implementation and development of practices most compatible with intensive agricultural production. Practices used in controlling erosion and sediment transport include the following: 1) cover crop maintenance, 2) strip cropping, 3) crop rotation, 4) contour farming, 5) mulching, 6) mulching with minimum tillage, 7) rough tillage to increase infiltration, and 8) terracing and other slope alteration practices. Maintenance of a cover crop or crop residue cover on the soil is recognized to be of major importance in soil stabilization and erosion control.

Soil fertilization increases the available nutrient supply for crop utilization and for transport in surface runoff (Taylor, 1967; Stanford et al., 1970; EPA, 1971). While instances of increased nutrient concentrations in runoff from fertilized soils have been reported (see Section 3.8), the impact of soil fertilization on nutrient transport in runoff is uncertain.

Aside from sediment transport, the movement of soil phosphorus in surface runoff should be related to the concentrations of soluble phosphorus maintained in the soil solution. Consequently, the forms and amounts of phosphorus applied, the timing of phosphorus applications in relation to expected periods of high runoff, and the incorporation of fertilizer phosphorus into the soil are important in minimizing phosphorus transport in runoff (Taylor, 1967; Martin et al., 1970).

The solubility of NO_3^- has focused major attention on the mobility of NO_3^- in surface runoff and percolating waters. Fertilization practices which minimize NO_3^- losses have been emphasized. Important practices include balancing application rate with crop needs, split applications and side-dressing to time applications with crop utilization, and use of deep-rooted crops in crop rotations to return leached NO_3^- to the soil surface (Martin et al., 1970; Stanford et al., 1970). Nitrification inhibitors may prove useful in retarding NO_3^- formation and thereby reducing losses.

Practices which may lead to increased transport of nutrients from agricultural soils include fall fertilizer application, broadcast application, and addition of fertilizers, including manure, to frozen soil (Biggar and Corey, 1969; Martin et al., 1970). Relatedly freezing apparently enhances the release of nutrients from crop plants, and this process may contribute substantially to nutrient transport during the spring runoff period (Holt, 1969).

The concern over water quality deterioration, the increasing costs and decreasing supplies of fertilizers, and decreasing food supplies emphasize the need for technology to maximize efficient fertilizer use and crop production while minimizing water pollution problems. Procedures for evaluating fertilizer application rates should not only focus on maximizing production but on minimizing fertilizer use and the potential for loss in runoff and seepage waters.

Section 6

NEED FOR NEW RESEARCH, DEMONSTRATION OR MONITORING PROGRAMS

6.1 Gaps in Knowledge Likely to be Remaining By the 1976 Time Frame

Accurate estimates fo the nutrient contribution of agricultural lands to the Great Lakes can not be made at the present time. The contribution is estimated to be about 20 percent of the total loading for phosphorus and will increase in proportion as point sources of phosphorus pollution are reduced through waste treatment. However, information on nutrient transport from agricultural land has been obtained from widely scattered locations and frequently under poorly defined conditions. Because of the lack of precision in estimates of nutrient transport in runoff, accurate evaluations of the role of land use and other factors in nutrient loadings can not be made.

The factors and mechanisms controlling the amounts of nutrients transported from agricultural lands are not established sufficiently to facilitate management programs. The physical and chemical factors controlling concentrations of soluble nutrients contained in surface runoff are not understood to the extent that quantitative predictions can be made. The effect of fertilizer use on soluble nutrient transport is uncertain.

Information is lacking on nutrient transport in streams and rivers. It is evident that a considerable portion of the soil eroded from agricultural lands is deposited prior to discharge into lakes. Furthermore, soluble nutrient loss from runoff waters through interaction with stream bank and bottom sediments can not be predicted.

The effect of particulate nutrients transported to lakes on the nutrient status of lake surface waters is poorly understood. Consequently, the validity of including particulate nutrients in lake nutrient budgets is uncertain. This arises because of a lack of information on the rates of release of nutrients from particulate forms in lake systems, the

availability of particulate forms to aquatic organisms, and the rates of nutrient uptake from particulate forms by algae as related to the longevity of particulate nutrient forms in lake surface waters.

6.2 New Studies Needed to Define the Problems Remaining

Gaps in knowledge regarding nutrient transport from agricultural lands to the Great Lakes (Section 6.1) indicates a need for research and demonstration projects in the areas discussed below to provide the information required.

6.2.1 Basic Research

Intensive investigation and monitoring of nutrient transport from representative agricultural watersheds. This type of investigation is needed to provide quantitative information on nutrient transport as a function of land use, soil type, fertilization, runoff and other important factors. Measurements should be made over a sufficient time period to provide a representative, accurate assessment. Relation to important events such as spring runoff should be evaluated to facilitate management and control programs.

Determination of the factors and mechanisms controlling the amounts of nutrients transported from agricultural lands. Research on the physical and chemical factors controlling the concentrations of soluble nutrients in runoff should be emphasized. The effects of soil properties, fertilizer application, and fertilization history of the soil should be established. This information would identify the specific processes and factors controlling soluble nutrient transport and, in turn, aid programs to control the problem.

Determination of the factors and processes controlling nutrient transport in streams and rivers. The extent of soluble nutrient removal by stream bank and bottom sediments should be established. This information would relate measurements of nutrient transport from agricultural land to the amounts expected to reach the Great Lakes.

Determination of the effect of particulate nutrients in runoff on the nutrient status of lake surface waters. Information is needed on a) the rate and extent of nutrient release from particulate forms, b) the biological availability of particulate nutrient forms and c) the rates of utilization of particulate nutrients as related to the rates of particulate nutrient removal from lake surface waters through settling. This information would greatly improve predicting of the impact of agricultural runoff on lake water quality.

6.2.2 Demonstration Projects

Monitoring of representative agricultural drainage basins. As discussed above (Section 6.2.1) this type of program is needed to provide quantitative information on nutrient transport from agricultural lands.

Effectiveness of sediment control through soil and water conservation programs in reducing the amounts of nutrients transported from agricultural lands. Considerable amounts of nutrients are transported with eroded soil. Control of soil erosion will also reduce the amounts of soluble nutrients transported from soil. Projects such as those described for Allen County, Indiana and Washington County, Wisconsin (See Section 3.9) are needed to determine whether soil conservation and sediment control programs are effective in reducing sufficiently the transport of nutrients from agricultural lands, and whether these programs can be implemented under existing legislation, policy and institutional arrangements.

6.2.3 Development of New Technology

It is likely that the basic technology exists to control nutrient transport from most agricultural lands through appropriate farming practices and soil erosion control programs. Furthermore, it is likely that technology can and will improve. However, the research and demonstration projects described above are needed to establish sufficiency of existing technology. If this technology is found to be adequate, the

major problem remaining is implementing the appropriate technology throughout the Basin. Since agriculture is likely to become more intensive to increase food production per unit area, fertilizer use technology should receive careful attention. Excessive fertilizer use will become less economical and may be detrimental to water quality. Methods of assessing fertilizer needs should be examined closely to ensure that these methods predict the appropriate time, rate, type and method of fertilizer application so that a high level of production can be maintained with minimal harmful effects on water quality.

Section 7

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Category A7 - Erosion and Sedimentation

ASSESSMENT OF EROSION AND SEDIMENTATION TO THE U. S. PORTION OF THE GREAT LAKES BASIN

prepared by
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GREAT LAKES BASIN COMMISSION
Ann Arbor, Michigan

To be used as portion of the U. S. Task A Report on GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by the International Joint Commission - prepared in partial fulfillment of U. S. Environmental Protection Agency Contract No. 68-01-1598

June 4, 1974

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Section I-1

INTRODUCTION

In this paper, sediment to the Lakes is the product of sheet and gully erosion of the soils from the agricultural and urban land use categories only. The larger sediment yield rates measured on the major streams in the basin are 87 metric tons per square kilometer (250 tons per square mile) from the Cuyahoga River at Independence, Ohio and 263 metric tons per square kilometer (750 tons per square mile) from the Genesee River in New York. While these sediment yields are moderately low in comparison to other parts of the U. S., sediment yields of this magnitude can cause serious problmes to harbors and fish. The sediment is a carrier of nutrient elements and pesticides. It also creates turbidity in the streams and the Great Lakes.

Erosion creates a wide range of sediment sizes ranging from coarse to fine grained material. Particles of all size fractions no doubt are contributed as sediment yield to the Lakes. The major portion of the sediment, however, is fine grained.

The most severe erosion occurs around the southern end of Lake Michigan and south of Lakes Erie and Ontario. Rivers and streams which drain this area reflect these higher erosion rates by a corresponding higher sediment yield than in other streams in the basin.

Section I-2

SUMMARY OF FINDINGS AND CONCLUSIONS

Sediment yield to the Lakes from erosion on agricultural and urban land from the U. S. portion of the Basin can be estimated with reasonable accuracy. Wise land use and effective use of soil conservation systems can effectively reduce or minimize these sedimentation problems. Conservation practices which are a part of these systems reduce runoff and erosion. If the conservation program for the future is of similar magnitude to that of the past, then the sediment yield to the Lakes is expected to remain about the same as present until 1980 and decline slightly below present levels in 2000. With present rates of land treatment by 2020 the sediment yield is expected to increase to about 6% above the present figure. This variation in yield is due mainly to an expected decline and then increased demand for row crops in the Basin. The general upward trend in yield is attributed to the projected continuing increase in areas undergoing urban construction. If the conservation program is increased above the present rates, then a significant decrease in sediment yield can be expected. The amount of decrease will be directly related to the amount of soil conservation practices which can be installed.

If the estimates developed in this paper are to be refined, new studies will be needed. These studies will be in the nature of research, data gathering, and an expanded water quality monitoring program.

The research needs are quite broad and cover items such as refinement of sediment delivery ratios for large watersheds, effect of land treatment on sediment yield from large watersheds and refinement of the factors influencing erosion and sediment yield from urban land. The data gathering should include remeasuring of sediment in reservoirs already measured and the measurement of new reservoirs. Expanding the water quality monitoring program would be necessary. The addition of more sediment gauging stations in the program for an adequate period of time would provide the data to predict sediment yield to the Lakes more accurately. These stations will also provide data on the effect of changes in land use on sediment yield.

Section I-3

RECOMMENDATIONS

Accurate and reliable estimates are needed on the sediment yield to the Lakes. Additional stream gauging stations where sediment is measured are required. Many more stations located near the junction of the stream and lake are needed to measure total suspended load from the streams flowing directly into the lake. Additional stations on the streams within the watershed are required to provide data with which sediment yield from different land uses can be determined.

One of the basic procedures used in sedimentation work utilizes sediment delivery ratio as a technique to reduce gross erosion to sediment yield. The sediment delivery ratio curve now in use was not developed for use on large watersheds. Research should be done to develop a new curve which is valid for large watersheds.

It is desirable to start a program to measure sediment in all reservoirs which meet the requirements for this type of operation. Reservoirs which have not been measured as well as those measured previously should be brought under this program. These data will help refine (1) sediment yield from large watersheds, (2) sediment delivery ratio, (3) the effect of land treatment on sediment yield on large watersheds, (4) sediment yield from and the effect of land treatment on soils undergoing urban construction and (5) sediment yield from settled urban areas.

Research should be intensified to determine the magnitude of sediment yield from urban land and from soils undergoing urban construction. Material gathered from this study can be used to refine procedures for predicting sediment yield from these types of areas.

Ordinances and regulations should be developed in each state on the control of erosion, runoff and sediment. These ordinances should be adequately funded so that their operation and enforcement is insured.

There should be a program which would offer the farmer and urban developer an incentive to apply sediment control measures. This could be in the form of cost-sharing or tax write-off.

Section II-1

INTRODUCTION

Pollution of the Great Lakes by sediment has been going on since the Lakes were formed. This sediment was from what is considered normal or geologic erosion. It was only when man manipulated the vegetation and land use for his own ends that the erosion on the land and sediment yield to the Lakes made a significant increase. Since studies of pollution to the Lakes began, it has been recognized that agricultural sediment, from the standpoint of bulk, is the major pollutant to the Lakes. However, to simply brand it as the most serious pollutant because of quantity may be an erroneous conclusion.

Sediment yield to the Great Lakes was estimated to be 453,900 metric tons(500,000 short tons) per year in the early 1800's. This yield has now increased to about 4,316,200 metric tons (4,757,800 short tons) and unless an intensive land treatment program is instituted the sediment yield is expected to increase to 4,557,100 metric tons (5,623,000 short tons) by 2020. Sediment yields of this magnitude, while larger than they should be for the intensity of development which exists, are not a serious problem to the Great Lakes as a whole. Serious local problems exist where streams discharge sediment into the Lakes. Sediment at these locations deposits in harbors, destroys fish habitat, reduces the recreational value of the lake, and increases municipal water treatment costs.

Section II-2

SCOPE OF STUDY

The details of this study cover sediment from sheet and gully erosion on agricultural land and urban areas but not from streambank erosion, forested land, transportation facilities, mining property, or recreation land. Information on sediment from these categories will be located in their respective reports. Originally the sediment assessment from urban land was to be the point where sediment entered sewers or man-made conveyances. This type of breakdown was not possible within the time allotted so this report includes all sediment from sheet and gully erosion on urban land. The material considered for this study covered a wide range, all the way from general reports on sedimentation to papers on specific subjects within the field.

There were several sources of data which were considered. They include files of the Soil Conservation Service, five Annotated Bibliographies on Hydrology and Sedimentation by the Water Resources Council or its predecessor plus one by the USGS, Catalog of Information on Water Data, two Inventories of Sediment-Load Data, a Summary of Reservoir Sediment Deposition Surveys, 15 issues of Notes on Sedimentation Activities, abstracts from the Smithsonian Science Information Exchange and from the Current Research Information System. Specific sources are detailed in Section II-3.

Sedimentation as a science is a relatively young discipline. Active interest in the field started in the early 1930's and has been growing ever since that time. It was just this year (1973) that the American Society of Testing Materials instituted a committee on this subject. Since there has been an increasing interest and study of this discipline, tremendous progress has been made in predictive techniques and measurement of sediment. Because of the dynamic progress that has overtaken the discipline, few papers or studies before 1950 were considered germane to this report. In most cases data developed before 1950 have been refined and expanded to the point where it would be redundant and possibly inappropriate to use the older material. Therefore, no material referring specifically to the Great Lakes was used which was dated before 1950. However, a book by H. H. Bennett of 1939 vintage was included in the general reference material.

Section II-3

STUDY PROCEDURE

The study was accomplished by first conducting a literature review. After this review contacts were made with various research personnel for interviews. The list of interviews does not necessarily represent all individuals working on these "categories" who could contribute data. It really represents those individuals which because of time and budget limitations could be contacted during the course of normal work.

Interviews were held with:

Ohio State University
- Agronomy Department
  - Dr. L. P. Wilding
  - Dr. T. Logan
- Agricultural Engineering Department
  - Dr. G. O. Schwab

University of Cincinnati
- Geology Department
  - Dr. P. E. Potter

Purdue University
- Agronomy Department
  - Dr. J. V. Mannering
  - Dr. D. Wiersma
- Agricultural Engineering Department
  - Dr. E. J. Monke

Agricultural Research Service
- W. H. Wischmeier, Research Statistician

Soil Conservation Service
- Lansing, Michigan
  - J. Thompson, Geologist
- Madison, Wisconsin
  - P. Cavanaugh, River Basin Party Leader
  - R. N. Cheetham, Jr., Geologist
- Columbus, Ohio
  - J. H. Harrington, Jr., State Conservation Engineer
- Indianapolis, Indiana
  - E. Pope, State Conservation Engineer
  - L. Kimberlin, State Resource Conservationist
  - C. Gossett, Geologist

Sources of literature which were reviewed are as follows:

1. Files of the Soil Conservation Service

2. Annotated Bibliography on Hydrology 1951-1954 and Sedimentation 1950-1954. Bulletin No. 7, Dec. 1955, Subcommittee on Hydrology and Sedimentation - Inter-Agency Committee on Water Resources.

3. Annotated Bibliography on Hydrology and Sedimentation 1959-1962 (U. S. and Canada). Bulletin No. 8, Sept. 1964, Subcommittees on Hydrology and Sedimentation - Inter-Agency Committee on Water Resources.

4. Annotated Bibliography on Hydrology and Sedimentation 1963-1965, Bulletin No. 9, June 1969, Hydrology and Sedimentation Subcommittee - Water Resources Council.

5. Annotated Bibliography on Hydrology and Sedimentation 1966-1968, Bulletin No. 10, July 1970, Hydrology and Sedimentation Subcommittee - Water Resources Council.

6. Annotated Bibliography on Hydrology and Sedimentation United States and Canada, 1955-1958. United States Geological Survey, Water Supply Paper 1546, 1962.

7. Annotated Bibliography on Sedimentation. Sedimentation Bulletin No. 2, Feb. 1950, Committee on Sedimentation - Federal Inter-Agency River Basin Committee.

8. Catalog of Information on Water Data, Water Resources Region (04) Great Lakes, Edition 1972. United States Geological Survey.

9. Current Research Information System, USDA.

10. Great Lakes Framework Study, 1970.

11. Inventory of Published and Unpublished Sediment-Load Data in the U. S. Sedimentation Bulletin No. 1, April 1949, Subcommittee on Sedimentation, Federal Inter-Agency River Basin Committee.

12. Inventory of Published and Unpublished Sediment-Load Data United States and Puerto Rico, 1950-1960. U. S. Geological Survey Water Supply Paper 1547, 1962.

13. Summary of Reservoir Sediment Deposition Surveys Made in the U. S. Through 1970. United States Department of Agriculture, Misc. Publication 1266, July 1973.

14. Notes on Sedimentation Activities (15 issues from 1953 to 1972) Sedimentation Committee - Water Resources Council.

15. Smithsonian Science Information Exchange, Inc.

16. Urban Hydrology - A Detailed Bibliography With Abstracts, U. S. Geological Survey Water Resources Investigations 3-72.

Section III-1

GENERAL DESCRIPTION OF LAND USE ACTIVITY

Sediment sources have been classified according to causative factors, eroding agent, location and other criteria. Thus, there are man-made (accelerated) and natural geologic erosion; water, wind, ice and gravity erosion; and erosion of the land surface (sheet-rill) or drainageway (gully-channel) erosion. For the Great Lakes Basin, the predominant sediment sources are from sheet-rill erosion of cultivated soils.

Agricultural land, because of type and intensity of use and because of the large area exposed and its susceptibility to erosion supplies the greatest amount of sediment yield to the Great Lakes. Approximately 80 percent of the sediment delivered to the Great Lakes comes from sheet, rill and gully erosion on agricultural and urban land. The remainder is from sheet and gully erosion on forest land, from road banks, stream banks, and municipal and industrial wastes.

The most widespread influence, other than on site land damage from the erosion process, is the downstream effect on water and related land resources. Runoff water carries the product of erosion, and other solid wastes, and deposits this product as sediment. Sometimes this product remains suspended in the downstream waters for prolonged periods and constitutes a detrimental element to the quality of water.

Section III-2

LENGTH OF TIME ACTIVITIES IN PROGRESS

Erosion and the resultant sediment yield are natural processes and as such have been continually active. Before settlement of the land, most of the streams in the basin probably ran clear except for periods of flood and high water. Accelerated erosion no doubt started in the middle 1800's with land settlement, land clearing, and with shifting much of this land into farms (22)[1/]. Land settlement combined with poor farming methods and urban construction practices started the trend which has continued until recently of an ever increasing sediment yield to the lakes.

The discussion of the magnitude of erosion and sediment yields to the Lakes during this early period is mostly conjecture because little data on this subject were obtained until fairly recently. Even today there are insufficient data to make more than estimates of the problem. However, the beginning of the accelerated erosion process could not have been before land settlement and the farm population by 1850 was such that had records of soil erosion and sediment yield been kept they certainly would have recorded a noticeable change from the period before 1800. This has been verified by Kemp using pollen variations in recent lake sediments (22).

Since the premise has been made that accelerated erosion and sediment yield to the Lakes is related to the amount of land used as cropland, the following table might be of interest (12).

Date	Cropland (U. S. Portion of the Basin)	
	(millions of hectares)	(millions of acres)
1800	-	-
1850	1.8	4.4
1920	5.3	13.0
1970	13.0	32.1

1/ Numbers in parentheses refer to references cited in section VII, Principal References.

Section III-3

TYPES AND NATURE OF POLLUTANTS ASSOCIATED WITH SEDIMENT

A large proportion of pesticides, plant nutrients, nuclear fallout and infectious particles leave the land and enter water bodies by attachment to sediment particles.

The amount of nitrogen, phosphorus, and potassium reaching surface waters that can be attributed to the fertilization of agricultural lands are yet unknown. Not all nutrients in waters come from fertilizers. Large amounts come from decomposition of soil organic matter and other sources. Little phosphorus can be lost from most soils except by erosion. Nitrogen may be in organic compounds carried by sediment, from decaying plants which might have gotten the nitrogen from fertilizers, or the atmosphere.

Levels of pesticides found to move from agricultural lands may be found in varying amounts in the runoff water and/or sediments. For example, studies by the Agricultural Research Service show that the amount of DDT residue in soils was related to the amount of DDT applied in previous years. The amount of DDT dissolved in runoff water was not detectable but significant levels were found adsorbed on surfaces of sediment particles.

Section III-4

STATE OF THE ART

There are several ways to determine the sediment yield of a watershed depending upon the physical features and the data available (30). Average annual sediment yields may be obtained from: (1) sediment load records; (2) gross erosion and sediment delivery ratios; (3) measured sediment accumulation; and (4) predictive equations.

Suspended load records.--The most reliable method of determining sediment yield from large watersheds. Suspended sediment transported by a stream may be measured by sampling. Water discharge can be determined by gaging at particular stream cross sections. Sediment yields may be estimated from these data. When field data are obtained in a proper manner for an appropriate period of time, the calculated average annual suspended sediment yield can be considered quite reliable.

There are literally hundreds of streams draining into the Great Lakes but unfortunately only a few have stations where suspended load is measured. Of the 1329 stations recorded only 60 have sediment data (10)(33). Of these 60, most of them, or 48, have a record of over 5 years but only 12 have a record over 10 years. Of those stations with over 10 years record only 3 have weekly or daily records. Table 1 is an inventory of all sediment stations on streams in the Great Lakes Basin showing the name of station, location, and type of information acquired (10)(33).

In order to estimate suspended load to the Lakes the stations should be as close to the mouth of the stream as technically feasible so as to keep unmeasured flows below the station to a minimum. Many of the above mentioned stations are not located at the mouths of streams but are well up in the watershed. They were not established for the primary purpose of estimating suspended sediment yield to the Lakes.

Suspended load is only part of the total load moving in a stream. The other portions are bed load and sediment attached to floating material. These items are considerably more difficult to measure. Bed load often is estimated as a percentage of suspended load.

Gross erosion and sediment delivery ratios --- This method has been used with success by the Soil Conservation Service for many years (30). It is well suited for estimating current sediment yields and predicting the effect of land treatment and other measures on future sediment yields. The estimate of sediment yield is made by use of the following equation:

$$Y = E(DR)$$

where Y = sediment yield (tons/unit area/year)
E = gross erosion (tons/unit area/year)
DR = sediment delivery ratio (DR less than 1)

The gross or total erosion is the summation of all the water erosion taking place. It is developed by the use of Universal Soil Loss Equation or other methods. The sediment delivery ratio is selected from various curves after considering the following: (1) type of sediment sources; (2) magnitude and proximity of sediment sources; (3) the transport system; (4) texture of eroded material; (5) depositional areas enroute from source to point of consideration; and (6) watershed characteristics. The product of gross erosion and sediment delivery ratio is the sediment yield.

While this procedure has worked well in the past, it has been developed and used on watersheds of less than 1,036 square kilometers (400 sq. miles). The delivery ratio curve is only plotted for watersheds up to 1,554 square kilometers (600 sq. miles). The use of this technique for catchments such as the Maumee River of about 3,315 square kilometers (1,280 sq. miles) and The Grande River of about 14,660 square kilometers (5,660 sq. miles) by extending the curve might be expanding the curve past its limits of accuracy. Also the soil loss equations were not developed for prediction on large areas of less than 1 percent slope. However, the data developed by this technique has checked reasonably well with the few suspended load stations.

Measured sediment accumulation -- The measured sediment accumulation in reservoirs of known age and history are excellent sources of data for establishing sediment yields (30). However, reservoir deposition and sediment yield are not synonymous. The amount of accumulated sediment must be divided by the reservoir's trap efficiency to obtain the sediment yield. This takes into account the amount of sediment that passed through the reservoir.

The sediment yield of an unmeasured watershed may be estimated from that of a measured watershed in an area where the topography, soils, and land use are similar. In order to directly transpose sediment yield data, the size of the drainage area of the surveyed reservoir should not be less than one-half nor more than twice that of the watershed under consideration. Beyond these limitations the annual sediment yield may be adjusted on the basis of curves and formulas after considering the same factors listed under the delivery ratio discussion.

Unfortunately only 52 reservoirs have been surveyed in the Great Lakes Basin and none of them have watersheds of a size comparable to the major stream systems (5). As in the discussion of suspended load the formulas and curves were developed for catchments of less than 1,036 square kilometers (400 sq. miles). Extrapolating this data to watersheds the size of major streams in the Great Lake Basin might be overextending the accuracy of the curves and formulas. Table 2 is a list of all reservoirs surveyed for sediment in the Great Lakes Basin showing name

of reservoir, location, date of survey, drainage area, storage capacity, and total and average sediment accumulation.

Predictive equations -- Predictive equations based on watershed parameters have been developed in some areas to estimate sediment yield (30). These equations express sediment yield as a function of a combination of several measurable, independent variables. The variables may be the size of the drainage area, annual runoff, watershed shape, relief-length ratio, average slope, and expression of the particle size of the surface soil and others.

Such equations are not numerous but, where developed, they can be used with the understanding that their application must be confined to the specific area they represent. All formulas of this type so far have been developed in the western United States. Since none of these formulas are known to have been developed which are applicable to the Great Lakes Basin this approach does not appear viable.

A review of the methods used to predict or determine sediment yield would be incomplete without a discussion of erosion and problems of determining its magnitude.

Investigations have identified the basic factors involved in the sheet erosion process to be rainfall, soil erodibility, slope length, slope gradient, kind and condition of cover and an erosion control factor such as contouring. These factors have been incorporated into equations that provide a means of obtaining a quantitative estimate of the amount of soil material moved by the sheet erosion process. The basic procedure and weaknesses as applied to the Great Lakes Basin were mentioned previously under gross erosion and sediment delivery ratios.

The factors listed above most susceptible to change by man are kind and condition of soil cover and slope length. When the kind or condition of the soil cover is improved such as using a cover crop rather than leaving the land bare or the slope length decreased by means of terraces less soil is eroded. A quantitative estimate of the change can be determined by use of the equations. Information obtained in this manner is normally from small catchments because the factors necessary for use in the equation are not usually available for the larger basins. Also, while we know that structural measures, which can effect slope length, can cause a reduction of erosion we do not have precise techniques for predicting the exact amount of this reduction from large areas.

Gullies usually follow rill erosion, beginning in slight depressions of the land surface where in time the concentrated flow may cut a considerable channel. The shape of the channel is generally influenced by the relative resistance of the soil or underlying rock.

Streambank erosion and bed degradation are influenced primarily by the bank materials and resistance of the channel bottom to the character and direction of flow. Removal of natural vegetation from streambanks increases bank erosion. Increases in peak flows or prolonged above normal flows due to changes on the watershed such as urbanization can increase bank erosion. The presence of coarse bed material that a stream cannot pick up during reduced flows tend to result in the banks being attacked by the flowing water.

The determination of loss by the various types of channel erosion can be made in several ways such as (1) a comparison of aerial photographs of different dates to determine the annual growth of channels; (2) re-running existing cross sections to determine the difference in total cross-sectional area; (3) the assembly of historical data in order to determine the average age of channels and their average annual growth; and (4) field studies to estimate the average annual rate of growth, either by lateral erosion or incision, in terms of volume per unit length of channel.

Again these techniques are more appropriate to small watersheds than to large basins as the computations could tend to be cumbersome and unwieldy for large areas.

Sediment delivery ratio is an important fluvial sediment concept that can be defined as the ratio of the amount of sediment carried out of a basin to the gross erosion within the basin. As has been mentioned before reliable techniques for determining sediment delivery ratio have not been developed for large basins.

Once the process of transportation of sediment starts interest is usually on the mechanical process of movement or entrainment. A facet of entrainment that is generally neglected is the chemical changes in the clay fraction that could or might take place (32).

These changes could markedly affect the ability of the clay to carry fertilizer elements. Biological alteration of inorganic compounds could also take place (32). Data of this type are generally lacking.

The state of the art of estimating quantities of erosion resulting from urban areas and from areas undergoing development is in its infancy. This is equally true of estimating how much of the eorded material goes downstream and how far.

The best approach to determining sediment yield from urban and urbanizing areas is to measure the sediment either as suspended load or in a reservoir. Unfortunately there are few gauging stations or reservoirs located on small watersheds where urban sediment can be measured.

Another method is to use the gross erosion and sediment delivery ratio technique. Watersheds under urban development usually are small enough that size is not a limitation. However, potential soil loss equations were not developed with urban criteria so modification of the equations could introduce an error.

A topic which has not been didcussed is the availability of soils information. This information is in Published Soil Surveys. Published Soil Surveys contain a wealth of data which is essential for making predictions on erosion, sediment yield and planning for conservation treatment. Soil series and phase, and land capability classification are but a few of the items available in the report which are useful to the individual working in this discipline.

The attached map of the Annual Status of Soil Surveys shows that mapping is completed or in progress on approximately 98% of the U. S. portion of the Basin and that emphasis for completion is given to the most intensively used areas. It also shows that mapping is less than 50% complete in about 60% of the Basin. For small detailed sedimentation studies where published soil surveys are not available personal interviews with the soil survey party leader can probably furnish sufficient data. It would not be practicable to interview all the individuals necessary for a large broad general study. In this case it would be most desirable to have Published Soil Surveys available. The latest estimate on completion of soil mapping in the Basin is 1995 with present funds, or 1990 with additional funds.

Soils information with a less intensive approach than that contained in the Soil Survey Report can be obtained from a General Soil Map. Such maps have been completed for all states in the U. S. portion of the Basin. General Soil Maps are also available for most counties.

Table 1 - Sampling Stations with Sediment Data

MAP NUMBER	MAP LETTER	OWDC NUMBER	AGENCY STATION NUMBER	STATION NAME	LATITUDE	LONGITUDE	STATE	COUNTY	SITE	PERIOD OF RECORD BEGAN	PERIOD OF RECORD DISCONTINUED	INTERRUPTED RECORD	SEDIMENT: CONCENTRATION (SUSPENDED)	SEDIMENT: PARTICLE SIZE (SUSPENDED)	SEDIMENT: PARTICLE SIZE (BED MATERIAL)	SEDIMENT: OTHER	AGENCY REPORTING
				GREAT LAKES REGION													
21	Y	50509	04201500	ROCKY R NR BEREA OHIO	412422	0815313	OH	035	STREAM	1964			4	8	8		GS
21	Y	50508	04200500	BLACK R AT ELYRIA OHIO	412250	0820615	OH	093	STREAM	1962			8	8	8		GS
21	Z	50517	04208000	CUYAHOGA R AT INDEPENDENCE OHIO	412344	0813745	OH	035	STREAM	1948			3	8	8		GS
21	AA	50523	04212500	ASHTABULA R NR ASHTABULA OHIO	415119	0804543	OH	007	STREAM	1965			8	8	8		GS
21	AA	50521	04212000	GRAND RIVER NR MADISON OHIO	414426	0810248	OH	085	STREAM	1965			8	8	8		GS
21	AA	50518	04209000	CHAGRIN R AT WILLOUGHBY OHIO	413751	0812413	OH	085	STREAM	1965			4	8	8		GS
21	AB	50524	04213000	CONNEAUT C AT CONNEAUT OHIO	415534	0803618	OH	007	STREAM	1965		*	8	8	8		GS
21	AC	50507	04199500	VERMILION R NR VERMILION OHIO	412255	0821900	OH	093	STREAM	1950		*	8	8	8		GS
21	AC	50506	04199000	HURON R AT MILAN OHIO	411800	0823630	OH	043	STREAM	1950		*	8	8	8		GS
22	B	51813	04100500	ELKHART R AT GOSHEN IND	413536	0855055	IN	039	STREAM	1963	1968		*	*			GS
22	O	54386	04142000	RIFLE R NR STERLING MI	440421	0840112	MI	011	STREAM	1966	1968		4	8			GS
22	Q	54389	04151500	CASS R AT FRANKENMUTH MICH	431950	0834525	MI	145	STREAM	1966			4	8			GS
22	T	54388	04144500	SHIAWASSEE R AT OWOSSO MICH	430010	0841115	MI	155	STREAM	1966			4	8			GS
22	W	54390	04159500	BLACK R NR FARGO MICH	430532	0823705	MI	147	STREAM	1966	1970		4	8			GS
22	X	54392	04164000	CLINTON R NR FRASER MICH	423440	0825700	MI	099	STREAM	1966	1970		4	8			GS
22	Y	55081	32903 04165700	DETROIT R AT DETROIT MICH	422050	0825731	MI	163	STREAM	1957			*				FPA
22	AB	54393	04176500	R RAISIN NR MONROE MICH	415750	0833155	MI	115	STREAM	1966			4	8			GS
22	AE	51814	04182000	ST MARYS R NR FORT WAYNE IND	405916	0850603	IN	003	STREAM	1953	1968		*	*			GS
22	AF	50883	04185000	TIFFIN R AT STRYKER OHIO	413015	0842550	OH	171	STREAM	1952		*	8	8	8		GS
22	AG	50887	04189000	BLANCHARD R NR FINDLAY OHIO	410321	0834117	OH	063	STREAM	1965			8	8	8		GS
22	AH	50889	04191500	AUGLAIZE R NR DEFIANCE OHIO	411415	0842357	OH	039	STREAM	1965			8	8	8		GS
22	AH	50885	04186500	AUGLAIZE R NR FT JENNINGS OHIO	405655	0841558	OH	137	STREAM	1965			8	8	8		GS
22	AI	50892	04193500	MAUMEE R AT WATERVILLE OHIO	413000	0834246	OH	095	STREAM	1950			3	8	8	*	GS
22	AK	50901	04197000	SANDUSKY R NR MEXICO OHIO	410239	0831142	OH	147	STREAM	1965		*	8	8	8		GS
22	AK	50900	04196800	TYMOCHTEE C AT CRAWFORD OHIO	405520	0832100	OH	175	STREAM	1965			8	8	8		GS
22	AK	50897	04196000	SANDUSKY R NR BUCYRUS OHIO	404813	0830024	OH	033	STREAM	1965			8	8	8		GS
23	K	51203	04063700	POPPLE R NR FENCE WIS	454550	0882750	WI	037	STREAM	1963			4	7			GS
23	K	51202	04061000	BRULE R NR FLORENCE WIS	455331	0881557	MI	071	STREAM	1964	1968		8				GS
23	K	51204	04066000	MENOMINEE R NR PEMBINE WIS	453525	0874635	MI	109	STREAM	1964			*				GS
23	K	54368	04062400	MICHIGAMME R NR WITCH LAKE MICH	461448	0880045	MI	043	STREAM	1964			*				GS
23	O	51205	04080000	LITTLE WOLF R AT ROYALTON WIS	442445	0885155	WI	135	STREAM	1963	1970		*				GS
23	O	56322	04081000	WAUPACA R NR WAUPACA WIS	441950	0885945	WI	135	STREAM	1963	1971		*				GS
23	Q	65484	04072750	LAWRENCE C NR WESTFIELD WIS	435352	0893452	WI	077	STREAM	1967			*	*	*		GS
23	Q	67735	04073500	FOX R AT BERLIN WIS	435715	0885710	WI	047	STREAM	1967			*				GS
23	R	67734	04085200	KEWAUNEE R NR KEWAUNEE WIS	442730	0873323	WI	061	STREAM	1968			5	8	8		GS
26	A	51206	04086000	SHEBOYGAN R AT SHEBOYGAN WIS	434425	0874535	WI	117	STREAM	1963			*				GS
26	B	51207	04087000	MILWAUKEE R AT MILWAUKEE WIS	430600	0875430	WI	079	STREAM	1963			5	8	8		GS
26	B	67732	04086200	E B MILWAUKEE R AT NEW FANE WIS	433301	0881118	WI	039	STREAM	1968			5	8	8		GS
26	B	67731	04086340	N B MILWAUKEE R NR FILLMORE WIS	432858	0880339	WI	131	STREAM	1968			5	8	8		GS
26	B	67730	04086360	MILWAUKEE R AT WAUBEKA WIS	432822	0875923	WI	089	STREAM	1968			*				GS

MAP NUMBER	MAP LETTER	OWDC NUMBER	AGENCY STATION NUMBER	STATION NAME	LATITUDE	LONGITUDE	STATE	COUNTY	SITE	PERIOD OF RECORD BEGAN	PERIOD OF RECORD DISCONTINUED	INTERRUPTED RECORD	SEDIMENT CONCENTRATION SUSPENDED	SEDIMENT PARTICLE SIZE SUSPENDED	SEDIMENT PARTICLE SIZE BED MATERIAL	SEDIMENT OTHER	AGENCY REPORTING
26	B	67718	40865	CEDAR R NR CEDARBURG WIS	431925	0875848	WI	089	STREAM	1968	1971		*				GS
26	C	73122	04087257	PIKE R NR RACINE WIS	423849	0875138	WI	059	STREAM	1971			5	8	8		GS
26	C	51209	04087240	ROOT R AT RACINE WIS	424505	0874925	WI	101	STREAM	1964			5	8	8		GS
26	C	51208	04087220	ROOT R NR FRANKLIN WIS	425225	0875945	WI	079	STREAM	1964			5	8	8		GS
27	A	51200	04025500	BOIS BRULE R AT BRULE WIS	463215	0913545	WI	031	STREAM	1964			8				GS
27	A	73123	04026005	BOIS BRULE R NR L SUPERIOR WIS	464220	0913607	WI	031	STREAM	1971			5	7	7		GS
27	A	67729	04026300	SIOUX R NR WASHBURN WIS	464120	0905702	WI	007	STREAM	1968			8				GS
27	A	57119	NE-8	NE NEMADJI R	463107	0922322	MN	017	STREAM	1967	1968					5	111
27	B	51201	04027000	BAD R NR ODANAH WIS	462915	0904145	WI	003	STREAM	1964			5				GS
29	A	54362	04001000	WASHINGTON C AT WINDIGO MICH	475523	0890842	MI	083	STREAM	1964			8		7		GS
29	A	63278	04014500	BAPTISM R NR BEAVER BAY MINN	472015	0911200	MN	075	STREAM	1962		*	*	*			GS
29	B	72871	SLB 0	SUPERIOR BAY LIFT BRIDGE			MN	137	STREAM	1971						5	111
29	D	63279	04018750	ST LOUIS R AT FORBES MINN	472148	0923556	MN	137	STREAM	1967	1971		5	7			GS
29	D	57120	SL-19.1	SAINT LOUIS R	463910	0921655	MN	137	STREAM	1953						5	111
29	D	57122	SL-120	SAINT LOUIS R	472147	0923550	MN	137	STREAM	1967	1968					5	111
29	D	57121	SL-51.9	SAINT LOUIS R	465050	0923434	MN	137	STREAM	1962						5	111
30	A	57123	LS-3	LAKE SUPERIOR	465154	0915730	MN	137	LAKE	1967	1968					5	111
30	A	57124	LS-2	LAKE SUPERIOR	471810	0911407	MN	075	STREAM	1967						5	111
30	A	57125	LS-1	LAKE SUPERIOR	474411	0902314	MN	031	LAKE	1967						5	111
30	B	55958	6373-496	LK MICHIGAN GREAT LAKES ILL	421847	0874809	IL	097	LAKE	1960			3				NFF

PART B – QUALITY OF SURFACE WATER

Stations in this Part are arranged in downstream order. That is, stations are listed in a downstream direction along the main stream, and stations on tributaries are listed between main-stream stations that are upstream and downstream from the mouth of the tributary. Stations on tributaries that enter above the first main-stream station are listed before the first main-stream station. Stations on tributaries to tributaries are listed in a similar manner.

Pages in this listing are arranged so that station identification, including period of record and storage of data, are on the left-hand pages, and the types of data available are on the facing right-hand pages. Following are brief explanations of the major column headings of the station listings in Part B:

Explanation of Major Column Headings in Part B

Map Number
Geographic unit and subunit in which station is located (see Fig. 1).

OWDC Number
This number, assigned by OWDC, is used to identify a specific data-acquisition activity included in the Catalog, and to crosstie the entry in the listing with the plot on the companion station-location maps.

Agency Station Number and Name
Assigned by reporting agency. To meet space limitations, the following abbreviations generally are used in the station names; in some instances, others may have been used:

AB	Above	MF	Middle Fork
BL	Below	NF	North Fork
B	Branch	SF	South Fork
EB	East Branch	WF	West Fork
MB	Middle Branch	L	Little
NB	North Branch	LK	Lake
SB	South Branch	M	Middle
WB	West Branch	N	North
BK	Brook	NR	Near
C	Creek	P	Pond
CA	Canal	R	River
CL	Coulee	RE	Reservoir
D	Ditch	RN	Run
DI	Distributary	S	South
E	East	STR	Stream
ES	Estuary	TR	Tributary
F	Fork	W	West
EF	East Fork		

Latitude-Longitude
As given by the reporting agency.

State and County
Symbols are used for States (and other areas), and code numbers are used for counties and independent cities in accordance with Federal Information Processing Standards (see Table 4).

Site
Type of water body for which data are acquired at the listed station: Stream, Canal, Lake, Reservoir (Reser), Estuary (Estary), Spring, Drain, Other.

Period of Record
Calendar year of beginning or ending of station activity.

Interrupted Record
Asterisk (*) indicates that during the period of record, data acquisition was suspended one or more times for an interval of one year or more.

Storage of Data
Asterisk (*) indicates in what form data are stored.

Types of Data
Frequency of determination or measurement of a parameter listed is indicated by the following numbers. If part or all of the data are telemetered, this is shown by the letter "T."

1. Continuous
2. Seasonal
3. Daily
4. Weekly
5. Monthly
6. Quarterly
7. Annual
8. Other periodic

An asterisk (*) indicates that the parameter has been measured, but the measurement was discontinued, or that frequency is unknown.

Supplementary data
Asterisks (*) are used in appropriate columns to indicate other types of water-data activities at the site.

Agency Reporting
Agency codes are given in Table 1.

III-4 Table 2 Reservoir Sediment Surveys in the U. S. Portion of the Great Lakes Basin[1/]

Reservoir	Nearest Town - State	Date of Survey	Net Drainage Area Sq Km	Net Drainage Area Sq Mi	Storage Capacity Cu M	Storage Capacity Ac Ft	Total Sediment Accum. Cu M	Total Sediment Accum. Ac Ft	Average Annual Sediment Accum. Cu M	Average Annual Sediment Accum. Ac Ft
1. Lake Rockwell	Kent, Ohio	Aug. 1914	321.4	(124.10)	9,160,000	(7,423.0)	-	-	-	-
ditto	ditto	Aug. 1950	ditto	ditto	8,499,000	(6,887.0)	661,400	(536.0)	18,400	(14.88)
2. Babb Pond	Richfield, Ohio	1932	0.05	(0.02)	302	(0.245)	-	-	-	-
ditto	ditto	Apr. 1951	ditto	ditto	233	(0.189)	69	(0.056)	4	(0.003)
3. Basom Pond	Hudson, Ohio	1944	0.83	(0.32)	4,780	(3.87)	-	-	-	-
ditto	ditto	Apr. 1951	ditto	ditto	4,000	(3.24)	780	(0.63)	120	(0.096)
4. Christener Pd.	Parma, Ohio	1940	0.23	(0.09)	4,200	(3.40)	-	-	-	-
ditto	ditto	Apr. 1951	ditto	ditto	3,440	(2.79)	750	(0.61)	68	(0.055)
5. Schoenbeck Pd.	Richfield, Ohio	1940	0.08	(0.03)	1,830	(1.48)	-	-	-	-
ditto	ditto	Apr. 1951	ditto	ditto	1,590	(1.29)	230	(0.19)	21	(0.017)
6. East Branch	Burton, Ohio	1939	43.72	(16.88)	5,749,000	(4,659.0)	-	-	-	-
ditto	ditto	Jun. 1949	ditto	ditto	5,596,000	(4,535.0)	153,000	(124.0)	15,300	(12.4)
7. Centerville Mills	Aurora, Ohio	1855	26.9	(10.38)	106,500	(86.3)	-	-	-	-
ditto	ditto	1949	ditto	ditto	47,300	(38.3)	54,300	(44.0)	580	(0.468)
8. Grand	Celina, Ohio	1844	240.9	(93.0)	160,636,000	(130,175)	-	-	-	-
ditto	ditto	Aug. 1940	ditto	ditto	131,551,000	(106,605)	29,085,000	(23,570.0)	303,000	(245.5)
9. Goller Pond	Defiance, Ohio	Mar. 1945	0.06	(0.024)	11,700	(9.5)	-	-	-	-
ditto	ditto	Aug. 1951	ditto	ditto	11,600	(9.4)	123	(0.1)	19	(0.015)
10. Auglaizer Power	Defiance, Ohio	1912	6,024	(2,326.0)	17,770,000	(14,400.0)	-	-	-	-
ditto	ditto	1951	ditto	ditto	14,314,000	(11,600.0)	3,455,200	(2,800.0)	88,500	(71.75)

1/ From Great Lakes Basin Framework Study, App. 18, Erosion and Sedimentation.

Table 2 continued

Reservoir	Nearest Town - State	Date of Survey	Net Drainage Area Sq Km	Net Drainage Area Sq Mi	Storage Capacity Cu M	Storage Capacity Ac Ft	Total Sediment Accum. Cu M	Total Sediment Accum. Ac Ft	Average Annual Sediment Accum. Cu M	Average Annual Sediment Accum. Ac Ft
11. Eagle Creek	Defiance, Ohio	1912	13.5	(5.2)	159,000	(129.0)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	91,300	(74.0)	67,870	(55.0)	1,740	(1.41)
12. Beetree Creek	Defiance, Ohio	1912	5.0	(1.91)	182,600	(148.0)	-	-	-	-
ditto	ditto	Aug. 1951	ditto	ditto	128,300	(104.0)	54,300	(44.0)	1,390	(1.13)
13. Batt Pond	Defiance, Ohio	Apr. 1947	0.03	(0.012)	3,200	(2.6)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	3,090	(2.5)	123	(0.1)	28	(0.023)
14. Harrison Lake	Fayette, Ohio	1941	96.0	(37.0)	1,223,000	(991.0)	-	-	-	-
ditto	ditto	Jun. 1949	ditto	ditto	1,146,500	(929.1)	76,400	(61.9)	9,220	(7.47)
ditto	ditto	Jul. 1951	ditto	ditto	1,113,500	(902.4)	33,000	(26.7)	15,700	(12.7)
15. Allmandinger Pd	Ohio City, Ohio	Jan. 1945	0.09	(0.035)	6,270	(5.08)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	5,810	(4.71)	457	(0.37)	68	(0.055)
16. Bucyrus #2	Bucyrus, Ohio	1919	7.2	(2.79)	298,600	(242.0)	-	-	-	-
ditto	ditto	Jun. 1949	ditto	ditto	269,000	(218.0)	29,620	(24.0)	988	(0.8)
17. Contris Pond	Lafayette	1947	0.34	(0.13)	11,350	(9.2)	-	-	-	-
ditto	ditto	1951	ditto	ditto	9,750	(7.9)	1,600	(1.3)	401	(0.325)
18. Sixmile Creek	Defiance, Ohio	1912	55.4	(21.4)	1,227,800	(995.0)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	858,900	(696.0)	369,000	(299.0)	9,470	(7.67)
19. Burt Lake	Oakwood, Ohio	Sep. 1948	1.9	(0.74)	72,800	(59.0)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	70,340	(57.0)	2,470	(2.0)	881	(0.714)
20. Kohart Pond	Grover Hill, Ohio	Sep. 1943	0.05	(0.019)	2,950	(2.4)	-	-	-	-
ditto	ditto	Jul. 1951	ditto	ditto	2,590	(2.1)	123	(0.1)	16	(0.0128)

Table 2 continued

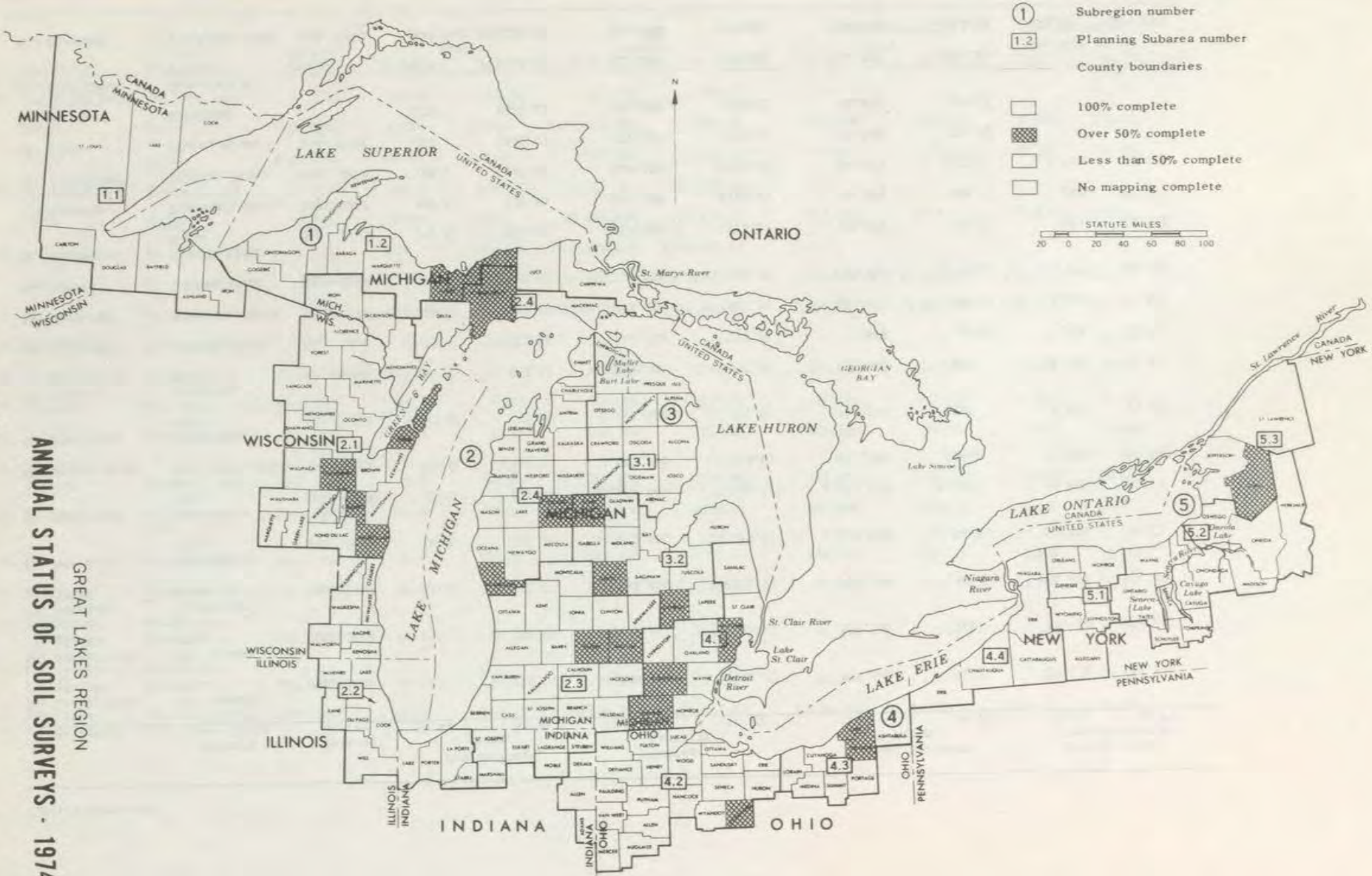
Reservoir	Nearest Town - State	Date of Survey	Net Drainage Area Sq Km	Net Drainage Area Sq Mi	Storage Capacity Cu M	Storage Capacity Ac Ft	Total Sediment Accum. Cu M	Total Sediment Accum. Ac Ft	Average Annual Sediment Accum. Cu M	Average Annual Sediment Accum. Ac Ft
21. Van Buren Lake	Findlay, Ohio	1939	58.8	(22.72)	306,000	(248.0)	-	-	-	-
ditto	ditto	Nov. 1948	ditto	ditto	253,000	(205.0)	53,100	(43.0)	5,580	(4.52)
ditto	ditto	Aug. 1951	ditto	ditto	229,500	(186.0)	23,400	(19.0)	8,370	(6.78)
22. Lake Rushford	Coneadea, N.Y.	1925	157.2	(60.7)	34,552,000	(28,000.0)	-	-	-	-
ditto	ditto	1951	ditto	ditto	33,844,000	(27,426.0)	708,300	(574.0)	27,270	(22.1)
23. Mount Morris	Mt. Morris, N.Y.	1951	2619	(1011.0)	417,104,000	(338,010.0)	-	-	-	-
ditto	Mt. Morris, N.Y.	1957	ditto	ditto	415,378,000	(336,611.0)	1,714,000	(1,389.0)	285,000	(231.0)
ditto	Mt. Morris, N.Y.	1963	ditto	ditto	413,875,000	(335,393.0)	3,106,000	(2,517.0)	269,000	(218.0)
24. Orchard Park	Buffalo, N.Y.		4.4	(1.7)	-	-	-	-	284	(0.23)
25. Saline Mill	Saline, Mich.	Mar. 1969	163.2	(63.0)	296,000	(240.1)	136,400	(110.5)	4,390	(3.56)
26. Bridgeway	Ann Arbor, Mich.	Mar. 1969	19.4	(7.5)	94,600	(76.7)	35,500	(28.8)	864	(0.70)
27. Franklin Mill	Franklin, Mich.	Apr. 1969	20.2	(7.8)	120,700	(97.8)	104,500	(84.7)	790	(0.64)
28. Tecumseh (Evans)	Tecumseh, Mich.	Apr. 1969	68.1	(26.3)	281,100	(227.8)	164,200	(133.1)	1,160	(0.94)
29. Sharon Hollow	Manchester, Mich.	May, 1969	64.8	(25.0)	318,500	(258.1)	140,900	(114.2)	3,330	(2.7)
30. Norvell	Norvell, Mich.	May, 1969	65.5	(25.3)	885,500	(717.6)	265,700	(215.3)	2,650	(2.15)
31. Brooklyn	Brooklyn, Mich.	May, 1969	16.1	(6.2)	307,600	(249.3)	77,700	(63)	3,700	(3.0)
32. Manchester (Power)	Manchester, Mich.	May, 1969	16.6	(6.4)	356,500	(288.9)	36,300	(29.4)	1,580	(1.28)
33. Manchester (Mill)	Manchester, Mich.	May, 1969	44.0	(17.0)	26,300	(21.3)	12,960	(10.5)	210	(0.17)
34. Kent Lake	Milford, Mich.	Jun. 1969	114.0	(44.0)	26,166,000	(21,204.0)	2,628,000	2,130)	136,600	(110.7)

Table 2 continued

Reservoir	Nearest Town - State	Date of Survey	Net Drainage Area		Storage Capacity		Total Sediment Accum.		Average Annual Sediment Accum.	
			Sq Km	Sq Mi	Cu M	Ac Ft	Cu M	Ac Ft	Cu M	Ac Ft
35. Stony Creek (North)	Mt. Vernon, Mich.	Jun. 1969	145.0	(56.0)	1,229,100	(996.0)	139,440	(113)	22,130	(17.93)
36. Stony Creek (South)	Mt. Vernon, Mich.	Jun. 1969	145.0	(56.0)	4,848,000	(3,929.0)	330,700	(268)	52,490	(42.54)
37. Oakwoods Metro.	Flat Rock, Mich.	Jun. 1969	81.8	(31.6)	1,161,400	(941.2)	372,000	(301.5)	8,450	(6.85)
38. Belleville	Belleville, Mich.	Jul. 1969	52.6	(20.3)	24,612,000	(19,945.0)	2,424,800	(1,965)	60,710	(49.2)
39. Ford Lake	Ypsilanti, Mich.	Jul. 1969	29.0	(11.2)	22,121,000	(17,926.0)	2,271,800	(1,841)	63,060	(51.1)
40. Barton Pond	Ann Arbor, Mich.	Jul. 1969	474.0	(183.0)	3,887,100	(3,150.0)	677,500	(549)	12,550	(10.17)
41. Iron Mill	Manchester, Mich.	Aug. 1969	13.5	(5.2)	1,913,900	(1,551.0)	485,000	(393)	4,850	(3.93)
42. Tecumseh Red (Mill)	Tecumseh, Mich.	Aug. 1969	67.1	(25.9)	835,400	(677.0)	420,800	(341)	4,210	(3.41)
43. H. N. Fry	Onsted, Mich.	Aug. 1969	32.4	(12.5)	149,700	(121.3)	6,540	(5.3)	938	(0.76)
44. Newburgh	Plymouth, Mich.	Sep. 1969	140.6	(54.3)	824,100	(667.8)	129,400	(104.9)	3,590	(2.91)
45. Adrian	Adrian, Mich.	Sep. 1969	152.8	(59.0)	1,234,000	(1,000.0)	183,900	(149.0)	6,570	(5.32)
46. Waterford	Northville, Mich.	Sep. 1969	140.0	(54.0)	213,480	(173.0)	88,800	(72.0)	888	(0.72)
47. Phoenix	Plymouth, Mich.	Sep. 1969	147.1	(56.8)	277,700	(225.0)	65,800	(53.3)	654	(0.53)
48. Fenton Mill	Fenton, Mich.	Jan. 1970	116.6	(45.0)	549,100	(445.0)	236,900	(192.0)	1,780	(1.44)
49. Elsie	Elsie, Mich.	Nov. 1964	497	(192.0)	137,000	(111.0)	72,700	(58.9)	-	-
50. Rockford	Rockford		583	(225.0)	109,800	(89.0)	55,000	(44.6)	-	-
51. Fish Creek	Carsonville, Mich.		319	(123.0)	122,200	(99.0)	36,200	(29.3)	-	-
52. Stronach	Wellston, Mich.	Jan. 1953	604	(233.0)	784,800	(640.0)	756,400	(613.0)	18,450	(14.95)

GREAT LAKES REGION

ANNUAL STATUS OF SOIL SURVEYS - 1974



Section III-5

MOBILITY OF SEDIMENT WITHIN SURFACE AND GROUND-WATER SYSTEMS

Erosion is caused by and sediment derived from the action of moving water, ice, or wind on rock and soil. Of the above agents moving water is predominant in the Great Lakes Region.

The transport of sediment occurs by two general processes, as a suspension in the water and as a bedload movement. When sediment is transported in suspension in the water it is referred to as "suspended load" and when it is moved along the bottom it is called "bedload." There are apparently no sharp divisions between the two forms of transport except that bedload moves close to the channel bottom whereas the suspended load is dispersed throughout the depth and width of the moving water. The physical mechanism of both forms of transport is very complex and it is beyond the purview of this report to explore its nature.

Generally, bedload movement involves the more coarse textured, heavy materials that require relatively high channel velocities. Suspended sediment movement occurs in moving water over a much wider range of velocities. In both forms of sediment transport, depth of flow, velocity of flow, and the nature of the material being transported are important factors in sediment transport rates.

The quantitative role that each of two broad forms play in the transport of sediment in the Great Lakes Basin is not known. Much evidence, visual observation, and logic leads to the conclusion that the majority of sediment transport occurs as suspended sediment. It is estimated that the quantity of bedload will not exceed 10 percent of the suspended load.

Sediment does not often get into the ground-water system. This is because the soil acts as a very effective filter and removes the sediment very soon after water begins to move through the soil. It is conceivable sediment laden water could get into the ground-water through fractures in rock or by overflowing into wells which penetrate fractured rock. It is not felt that this condition is prevalent.

Section III-6

NATURAL RENOVATION MECHANISM AVAILABLE TO REMOVE POLLUTANTS AND CONTAMINANTS (PHYSICAL, CHEMICAL, BIOLOGICAL)

Natural renovation mechanisms that remove sediment which are different from the natural process now utilized in soil conservation practices are limited. The most common and effective mechanism used is vegetation. Given sufficient plant cover soil erosion can be held to a minimum. Many of the soil conservation practices listed in Section V are based on this principle.

The effectiveness of vegetation as an erosion retardant varies widely. For example, erosion rates on a field can be reduced by 60% by converting from continuous corn to a rotation of two years corn, one year small grain, and one year grass and clover meadow. Conversion to continuous pasture would reduce erosion even more while conversion from close growing crops to pasture would reduce erosion at a lesser rate. Another example might be exclusion or management of livestock to allow the natural plant growth to regain adequate cover to protect the soil.

The other soil conservation practices listed in Section V also were developed to take advantage of natural processes. Debris basins and reservoirs are constructed to exploit the fact that gravity causes particles in suspension in the water to settle in the reservoir. Terraces and diversions reduce slope lengths. Minimum tillage, chiseling, incorporating residues into the soil and other such practices increase infiltration and decrease runoff. Other management, land treatment and structural measures utilize some natural physical process to effect a reduction in erosion or sediment yield. The effectiveness of these practices also varies widely. A reservoir can be from 100% effective to nearly ineffective depending on the reservoir trap efficiency. The other practices vary almost as much in effectiveness.

Both chemical and biological actions can take place which cause either retention of soils in place or deposition of sediment enroute. Vegetative growth which protects soils can be stimulated by fertilizers. Better soil structure which promotes infilgration and reduced runoff can be developed by the incorporation of crop residues into the soil. This interaction of the physical, chemical, and biological actions is extremely complex and in many cases more than one process is involved in reducing erosion and sediment yield. More often than not chemical and biological actions complement physical actions in the incorporation of conservation practices on the land.

Section III-7

PROBABLE CHANGES IN SEDIMENT YIELD IN TIME

For this section it was thought that the predicted sediment yields to the Great Lakes for specified years from agricultural and urban land would be informative. Data of this type were not available so it was developed in the following manner:

Data were available on the sediment yield from sheet erosion to each one of the Great Lakes (8). Since this figure included both forest and agricultural land the contribution from forest land needed to be deleted to leave only sheet erosion from agricultural land. This was done by applying a weighted factor which consisted of a combination of the erosion rates for each land use and amount of each land use. This factor times the sheet erosion figure gave the 1970 sediment yield from sheet erosion on agricultural land. Then a factor of 3 percent for gully erosion was added to the sheet erosion sediment yield figure to obtain the sediment yield from agricultural land from sheet and gully erosion for 1970.

To establish the sediment yields for future dates, the present sediment yield was multiplied by the percent change reflected in the Gross Erosion for Projected Dates (5). This table takes into account increased or decreased demand for different types of crops, projected advances in technology, displacement of agricultural land by urban growth, and the effect of an increased amount of land treatment. These projected changes are based on material regarding population and food and fiber requirements furnished by the Office of Business Economics (now the Bureau of Economic Analysis) - Economics Research Service and used in the Great Lakes Basin Framework Study (5).

The 1973 figures were obtained by interpolation of the 1970 and 1980 figures.

Table 3 through 7 of this section show Estimated Annual Sediment Yield from Sheet and Gully Erosion on Agricultural Land for the years 1976, 1980, 2000, and 2020.

The figures for years 1980, 2000, 2020 on Table 8, Estimated Average Annual Sediment Yield from Sheet and Gully Erosion on Urban Land to the Great Lakes, were determined by calculating a sediment delivery rate for each metropolitan area and multiplying it by the estimated erosion rate. The erosion rates were based on projected urban growth. The figure for 1973 was obtained from a direct proportion of the 1970 and 1980 erosion rate time the delivery rate for each metropolitan area.

TABLE 3

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE SUPERIOR [1/] [2/]

(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Superior Slope Complex	9.5	(10.5)	9.8	(10.7)	9.2	(10.1)	9.2	(10.1)
St. Louis River	2.5	(2.8)	2.6	(2.9)	2.4	(2.7)	2.4	(2.7)
Nemadji River	1.6	(1.8)	1.7	(1.9)	1.5	(1.7)	1.5	(1.7)
Apostle Complex	10.2	(11.2)	10.3	(11.4)	9.8	(10.8)	9.8	(10.8)
Bad River	3.4	(3.7)	3.4	(3.8)	3.3	(3.6)	3.3	(3.6)
Montreal River	1.0	(1.1)	1.0	(1.1)	1.0	(1.1)	1.0	(1.1)
Porcupine Mt. Complex	3.2	(3.5)	3.1	(3.4)	3.0	(3.3)	3.0	(3.3)
Ontonagon River	0.8	(0.9)	0.8	(0.9)	0.7	(0.8)	0.7	(0.8)
Keweenaw Pen. Complex	4.2	(4.6)	4.1	(4.5)	3.9	(4.3)	3.9	(4.3)
Sturgeon River	0.6	(0.7)	0.6	(0.7)	0.5	(0.6)	0.5	(0.6)
Huron Mt. Complex	1.8	(2.0)	1.7	(1.9)	1.6	(1.8)	1.6	(1.8)
Grand Marais Complex	2.2	(2.4)	2.1	(2.3)	2.0	(2.2)	2.0	(2.2)
Tahquamenon River	0.4	(0.4)	0.4	(0.4)	0.4	(0.4)	0.4	(0.4)
Sault Complex	0.5	(0.6)	0.5	(0.6)	0.5	(0.5)	0.5	(0.5)
TOTAL	41.9	(46.2)	42.1	(46.5)	39.8	(43.9)	39.8	(43.9)

[1/] All figures rounded to nearest 100 tons

[2/] Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 4

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE MICHIGAN 1/ 2/

(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Seul Choix Groscap	1.9	(2.1)	1.8	(2.0)	1.5	(1.7)	1.8	(2.0)
Manistique River	0.6	(0.7)	0.6	(0.7)	0.5	(0.6)	0.6	(0.7)
Bay De Noc	6.7	(7.4)	6.4	(7.0)	5.5	(6.1)	6.6	(7.3)
Escanaba River	4.4	(4.8)	4.2	(4.6)	3.6	(4.0)	4.4	(4.8)
Menominee Complex	4.4	(4.9)	4.3	(4.7)	3.7	(4.1)	4.4	(4.9)
Menominee River	19.9	(21.9)	19.0	(20.9)	16.6	(18.3)	19.7	(21.7)
Peshtigo River	4.2	(4.6)	4.0	(4.4)	3.5	(3.9)	4.2	(4.6)
Oconto River	26.2	(28.9)	25.0	(27.6)	22.0	(24.0)	26.0	(28.7)
Saumico Complex	33.9	(37.4)	32.3	(35.6)	28.3	(31.2)	33.7	(37.1)
Fox River	80.9	(89.2)	77.2	(85.1)	67.7	(74.6)	80.5	(88.7)
Green Bay Complex	127.8	(140.9)	121.9	(134.4)	106.8	(117.7)	127.0	(140.0)
Chicago-Milwaukee Complex	284.9	(314.1)	263.6	(290.6)	220.6	(243.2)	187.3	(206.5)
St. Joseph River	207.5	(228.7)	207.8	(229.1)	204.7	(225.6)	248.5	(273.9)
Kalamazoo River	187.5	(206.7)	187.8	(207.0)	185.0	(203.9)	224.5	(247.5)
Grand River	94.6	(104.3)	81.7	(90.1)	68.5	(75.5)	84.4	(93.0)
Black River	25.7	(28.3)	22.1	(24.4)	18.6	(20.5)	22.9	(25.2)
Muskegon River	46.1	(50.8)	39.8	(43.9)	33.4	(36.8)	41.1	(45.3)
Sable Complex	9.4	(10.8)	8.2	(9.0)	6.8	(7.5)	8.4	(9.3)
Manistee River	20.0	(22.0)	17.2	(19.0)	14.4	(15.9)	17.8	(19.6)
Traverse Complex	70.6	(77.8)	61.0	(67.2)	51.2	(56.4)	63.0	(69.4)
TOTAL	1,257.2	(1,385.9)	1,185.9	(1,307.3)	1,062.9	(1,171.7)	1,206.8	(1,330.2)

1/ All figures rounded to nearest 100 tons

2/ Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 5

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE HURON [1/] [2/]

(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Cheboygan River	1.4	(1.5)	1.4	(1.5)	1.2	(1.3)	1.2	(1.3)
Presque Isle Complex	2.4	(2.7)	2.4	(2.6)	2.1	(2.3)	2.1	(2.3)
Thunder Bay River	5.0	(5.5)	4.9	(5.4)	4.3	(4.7)	4.3	(4.7)
Alcona Complex	1.6	(1.8)	1.6	(1.8)	1.4	(1.5)	1.4	(1.5)
AuSable River	3.3	(3.6)	3.2	(3.5)	2.8	(3.1)	2.8	(3.1)
Rifle-AuGres Complex	12.9	(14.2)	12.7	(14.0)	11.2	(12.3)	11.2	(12.3)
Kawkawlin Complex	16.2	(17.9)	16.4	(18.1)	10.0	(11.0)	12.3	(13.6)
Saginaw River	115.2	(12.7)	116.6	(128.5)	70.9	(78.1)	87.5	(96.4)
Thumb Complex	60.2	(66.4)	61.0	(67.2)	37.0	(40.8)	45.7	(50.4)
TOTAL	218.2	(240.6)	220.2	(242.6)	140.9	(155.1)	168.5	(185.6)

[1/] All figures rounded to nearest 100 tons

[2/] Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 6

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE ERIE 1/ 2/
(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Black River	29.3	(32.3)	28.3	(31.2)	24.8	(27.3)	24.6	(27.1)
St. Clair Complex	25.1	(27.7)	24.2	(26.7)	21.1	(23.3)	21.0	(23.2)
Clinton River	41.7	(46.0)	40.3	(44.4)	35.2	(38.8)	34.9	(38.5)
Rouge Complex	33.8	(37.3)	32.7	(36.0)	28.5	(31.4)	28.3	(31.2)
Huron River	57.1	(62.9)	55.1	(60.7)	48.1	(53.0)	47.7	(52.6)
Raisin River	91.5	(100.9)	88.4	(97.4)	77.2	(85.1)	76.7	(84.5)
Maumee River	1,092.8	(1,204.6)	1,120.5	(1,235.1)	1,157.7	(1,276.1)	1,215.7	(1,340.1)
Toussaint-Portage Complex	108.6	(119.7)	111.3	(122.7)	115.0	(126.8)	120.8	(133.2)
Sandusky River	109.7	(120.9)	215.5	(237.6)	222.7	(245.5)	233.9	(257.8)
Huron-Vermillian Complex	138.7	(152.9)	142.2	(156.7)	146.9	(161.9)	154.3	(170.1)
Black-Rocky Complex	89.4	(98.5)	80.1	(88.3)	81.8	(90.2)	89.1	(98.2)
Cuyahoga River	86.5	(95.3)	77.5	(85.4)	79.2	(87.3)	86.2	(95.0)
Grand River	179.8	(198.2)	161.1	(177.6)	164.7	(181.5)	179.2	(197.5)
Ashtabula-Conneaut Complex	11.3	(12.5)	10.2	(11.2)	10.3	(11.4)	11.2	(12.4)
Erie-Chautauqua Complex	7.1	(7.8)	6.5	(7.2)	5.0	(5.5)	5.0	(5.5)
Cattaraugus River	14.1	(15.5)	13.1	(14.4)	9.9	(10.9)	10.0	(11.0)
Tonawanda Complex	23.0	(25.3)	21.3	(23.5)	16.1	(17.8)	16.2	(17.9)
TOTAL	2,139.5	(2,358.3)	2,228.3	(2,456.1)	2,244.2	(2,473.8)	2,354.8	(2,595.8)

1/ All figures rounded to nearest 100 tons

2/ Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 7

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL LAND (U.S. PORTION) TO LAKE ONTARIO [1/] [2/]

(TONS X 1000)

UNIT	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Niagara-Orleans Complex	48.7	(53.7)	43.9	(48.4)	41.6	(45.9)	41.6	(45.9)
Genesee River	59.7	(65.8)	53.9	(59.4)	51.1	(56.3)	51.1	(56.3)
Wayne-Cayuga Complex	49.5	(54.6)	44.4	(48.9)	31.2	(34.4)	36.6	(40.3)
Oswego River	99.4	(109.6)	89.2	(98.3)	62.7	(69.1)	73.4	(80.9)
Salmon-Perch Complex	43.0	(47.4)	38.6	(42.5)	27.0	(29.9)	31.8	(35.0)
Black River	24.5	(27.0)	22.0	(24.3)	20.7	(22.8)	20.7	(22.8)
Oswegatchie River	17.5	(19.3)	15.8	(17.4)	14.8	(16.3)	14.8	(16.3)
Grass-Raquette-St. Regis Complex	20.5	(22.6)	18.5	(20.4)	17.3	(19.1)	17.3	(19.1)
TOTAL	362.8	(400.0)	326.3	(359.6)	266.4	(293.8)	287.3	(316.6)

[1/] All figures rounded to nearest 100 tons

[2/] Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

TABLE 8

ESTIMATED AVERAGE ANNUAL SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON URBAN LAND (U.S. PORTION) TO THE GREAT LAKES 1/ 2/

(TONS X 1000)

METROPOLITAN COMPLEX	YEAR							
	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Lake Superior								
Duluth-Superior	13.8	(15.2)	14.2	(15.6)	16.0	(17.6)	18.0	(19.8)
SUB TOTAL	13.8	(15.2)	14.2	(15.6)	16.0	(17.6)	18.0	(19.8)
Lake Michigan								
Green Bay-Oskash	1.7	(1.9)	1.8	(2.0)	2.3	(2.5)	2.9	(3.2)
Milwaukee	3.6	(4.0)	4.0	(4.4)	5.4	(5.9)	7.1	(7.8)
Chicago	28.7	(31.6)	30.9	(34.1)	38.1	(42.0)	47.0	(51.8)
South Bend-Elkhart	5.3	(5.8)	5.5	(6.1)	6.7	(7.4)	8.3	(9.1)
Kalamazoo-Battle Creek	7.9	(8.7)	8.8	(9.7)	12.0	(13.2)	15.9	(17.5)
Grand Rapids	2.6	(2.9)	2.9	(3.2)	3.6	(4.0)	4.5	(5.0)
Lansing-Jackson	2.7	(3.0)	3.0	(3.3)	4.1	(4.5)	5.4	(6.0)
SUB TOTAL	52.5	(57.9)	56.9	(62.8)	72.2	(79.5)	91.1	(100.4)
Lake Huron								
Bay City-Saginaw-Flint	5.4	(5.9)	5.8	(6.4)	8.3	(9.1)	9.4	(10.4)
SUB TOTAL	5.4	(5.9)	5.8	(6.4)	8.3	(9.1)	9.4	(10.4)
Lake Erie								
Detroit	50.5	(55.7)	55.7	(61.4)	71.1	(78.4)	91.4	(100.8)
Toledo	5.1	(5.6)	5.8	(6.4)	6.7	(7.4)	8.4	(9.3)
Fort Wayne	6.5	(7.2)	7.3	(8.0)	10.0	(11.0)	13.6	(15.0)
Loraine-Elyria	11.3	(12.4)	12.1	(13.3)	15.1	(16.6)	18.7	(20.6)
Cleveland-Akron	102.4	(112.9)	109.4	(120.6)	136.5	(150.5)	170.0	(187.4)
Erie	9.2	(10.1)	9.7	(10.7)	11.5	(12.8)	14.3	(15.6)
SUB TOTAL	185.0	(203.9)	200.0	(220.4)	250.9	(276.7)	316.4	(348.7)

YEAR

METROPOLITAN COMPLEX	1973		1980		2000		2020	
	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons	Metric tons	Short tons
Lake Ontario								
Buffalo	19.1	(21.0)	20.3	(22.4)	24.6	(27.1)	29.7	(32.7)
Rochester	4.4	(4.8)	4.7	(5.2)	6.1	(6.7)	7.7	(8.5)
Syracuse-Rome	16.4	(18.1)	17.7	(19.5)	22.2	(24.5)	27.6	(30.4)
SUB TOTAL	39.9	(43.9)	42.7	(47.1)	52.9	(58.3)	65.0	(71.6)
TOTAL	296.6	(326.8)	319.6	(352.3)	400.3	(441.2)	499.9	(550.9)

1/ All figures rounded to nearest 100 tons

2/ Adapted from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, October 1970

Section III-8

PROJECTED SERIOUSNESS OF FUTURE POLLUTION PROBLEMS DERIVED FROM THESE LAND USE ACTIVITIES

In an effort to evaluate the problem, Table 9 and Figures 1 through 6 were developed. The Table and these Figures were developed from Tables 3 through 8 in Section III-4. Tables 3 through 8 are based on material from Appendix 18, Erosion and Sedimentation, Great Lakes Basin Framework Study, which is in turn based on projections used in the Framework Study and furnished by the Office of Business Economics - Economic Research Service. The projections consider changes in demand for different crops, advances in technology, displacement of agricultural land by urban growth and the effect of an increased amount of land treatment.

The total sediment yield from agricultural and urban land from the U. S. portion of the Basin to the Lakes for the last 40 years is about the same as the total for the 120 years before that period, truly a startling increase. On the other hand, the existing conservation programs are estimated to have reduced the potential sediment yield by about 23.5 million metric tons (26 million short tons) and will provide for an additional reduction of 58 million metric tons (64 million short tons) by 2020. In spite of the reductions directly attributable to conservation programs sediment yield from these sources is expected to increase approximately 6% by 2020.

Sediment yield from agricultural land is projected to decrease until 2000 and then increase. Unlike the composite figure (weighted total of agricultural and urban sediment) mentioned above, sediment yield from agricultural land is expected to be less than 1% higher in 2020 than at present. Sediment yield from urban lands is an ever increasing figure.

It is approximately 69% higher in 2020 than at present. The decline in sediment yield from agricultural land in 2000 is mostly due to a projected decrease in row crops throughout the Basin and an increase in close growing crops which provide better erosion protection for the soils. After 2000 this trend reverses with the result that at 2020 the sediment yield is slightly higher than the present figure. The increase in sediment yield from urban land is due to a projected constant increase in urban construction.

The greatest increase in sediment yield from agricultural and urban land from the U. S. portion of the Basin is approximately 15% to Lake Erie in 2020. This is largely because of a projected increase in row crops and a sizeable increase in urban construction. Figure 5 shows the composite sediment yield change to be continually increasing to 2020.

The largest decrease in sediment yield from these sources in the U. S. portion of the Basin is approximately 21% to Lake Huron in 2020. This is due to a projected decrease in row crops and an increase in close growing crops. The urban built-up land in the area is projected to increase a significant amount but this source is more than counterbalanced by the agricultural land change.

By 2020 the total sediment yield from these sources from the U. S. portion of the Basin to all the Lakes is expected to show a slight increase. Lake Superior is projected to have a slight increase, Lake Michigan a slight decrease, Lake Huron a significant decrease, Lake Erie a moderate increase, and Lake Ontario a moderate decrease. Lake Erie then is the only lake where the trend of increasing sediment yields from agricultural and urban areas from the U. S. portion of the Basin is projected to continue. The other lakes will have sediment yields from these sources at about the same level or less.

It is not known exactly what effect these changes in sediment yield will have on nutrients and pesticides which enter the Lakes. On the surface, it might appear that if the sediment yield changed then the amount of other contaminants might change in the same direction. While this premise might be correct it is also conceivable that increased or decreased use of these pollutants might influence the amount furnished to the Lakes more than changes of sediment yield. It is not within the scope of this category to make predictions on future use of pesticides and nutrients. Categories A5 and A6 of Task A cover the interrelationship of these pollutants with sediment in greater detail.

Table 9 shows an indication of future pollution problems from sediment to the Great Lakes compared to the present. However, this evaluation means little unless some understanding of the magnitude of the present problem is known. The larger sediment yield rates measured on major streams in the basin are 87 metric tons per square kilometer (250 tons per square mile) from the Cuyahoga River at Independence, Ohio and 263 metric tons per square kilometers (750 tons per square mile) from the Genesee River in New York. These sediment yields are moderately low in comparison to other parts of the U. S.

While the effect of yields of this magnitude on the Great Lakes is not large the yield from some streams can have a serious impact on that portion of the lakes near the junction of the stream and lake. The Maumee River discharges in excess of 1,070,300 metric tons (1,179,000 short tons) of sediment per year into Lake Erie. A significnat portion of this sediment is deposited close to shore. Toledo Harbor, at the mouth of the Maumee River, receives the second largest amount of annual maintenance dredging of the Great Lakes harbors (13). This dredging amounts to approximately 680,850 metric tons (750,000 short tons) or about 64% of the sediment yield of the river.

In addition to choking the harbor, this sediment serves as a carrier for nutrient elements, destroys fish habitat, reduces the recreational value of the lake, and increases municipal water treatment costs.

It can be seen that while small changes as shown on Table 9 will have little effect on the Great Lakes as a whole, they could have significant impact on local problem areas within the Lakes. Major changes in sediment yield might show an effect in the Great Lakes and they certainly would be felt in local problem areas.

Table 9

ESTIMATED PERCENTAGE CHANGE FROM 1973 OF SEDIMENT YIELD FROM SHEET AND GULLY EROSION ON AGRICULTURAL AND URBAN LAND (U.S. PORTION) TO THE GREAT LAKES

	Year		
	1980	2000	2020
All Lakes			
Agricultural Land	- 0.4	- 6.6	+ 0.9
Urban Land	+ 7.8	+35.0	+68.6
Composite	+ 0.1	- 3.8	+ 5.6
Lake Superior			
Agricultural Land	+ 0.7	- 5.0	- 5.0
Urban Land	+ 2.6	+15.8	+30.3
Composite	+ 1.1	+ 0.2	+ 3.8
Lake Michigan			
Agricultural Land	- 5.7	-15.5	- 4.0
Urban Land	+ 8.5	+37.3	+73.4
Composite	- 5.1	-13.3	- 0.9
Lake Huron			
Agricultural Land	+ 0.8	-35.5	-22.9
Urban Land	+ 8.5	+54.2	+76.3
Composite	+ 1.0	-33.4	-20.5
Lake Erie			
Agricultural Land	+ 4.1	+ 4.9	+10.1
Urban Land	+ 8.1	+35.7	+71.0
Composite	+ 4.5	+ 7.3	+14.9
Lake Ontario			
Agricultural Land	-10.1	-26.6	-20.9
Urban Land	+ 7.3	+32.8	+63.1
Composite	- 8.3	-20.7	-12.6

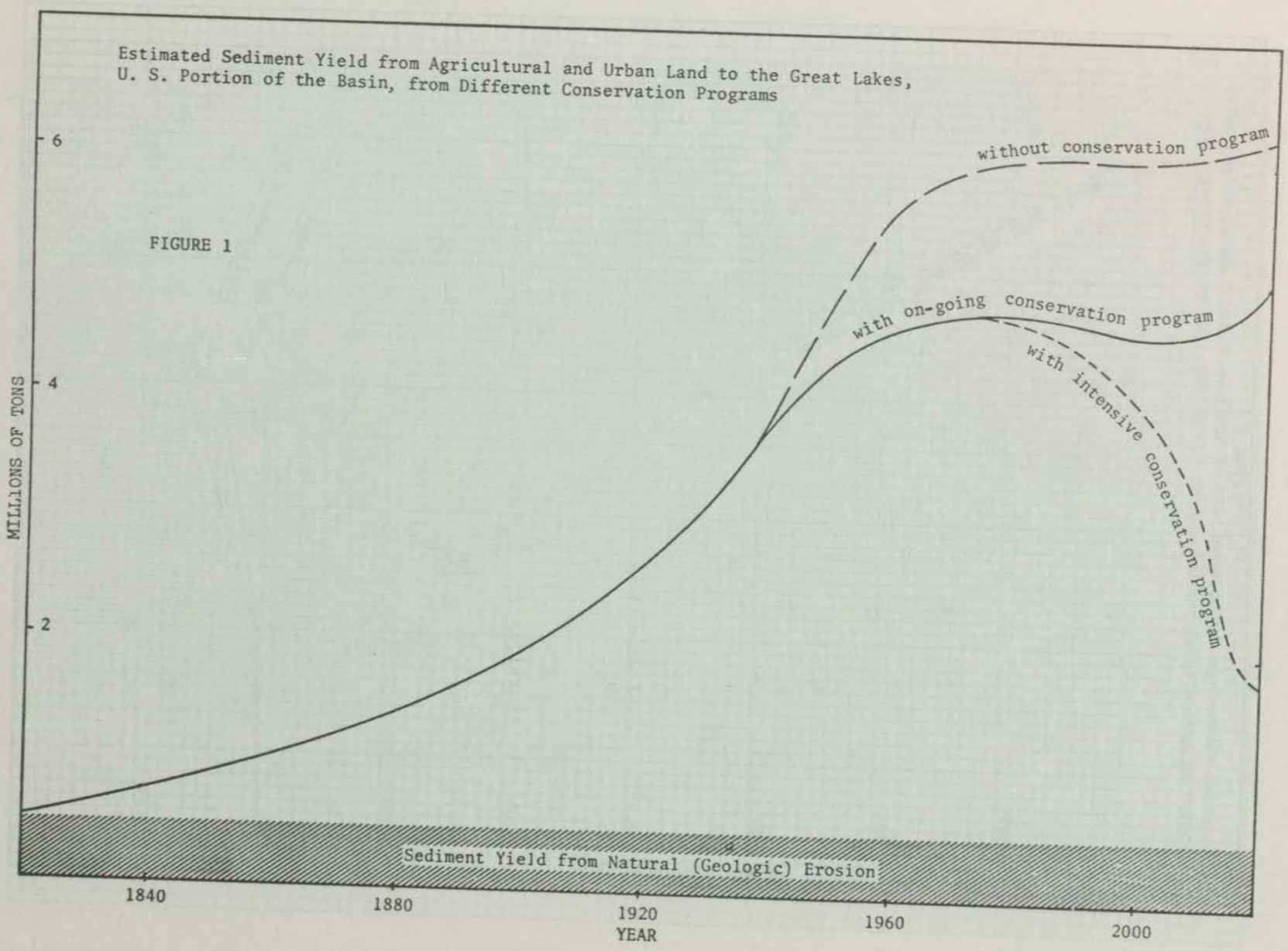

Estimated Sediment Yield from Agricultural and Urban Land to the Great Lakes,
U. S. Portion of the Basin, from Different Conservation Programs
FIGURE 1
MILLIONS OF TONS
6
4
2
without conservation program
with on-going conservation program
with intensive conservation program
Sediment Yield from Natural (Geologic) Erosion
1840
1880
1920
1960
2000
YEAR

FIGURE 2

Estimated Percentage Change From 1973 of Sediment Yield From Sheet and Gully Erosion on Agricultural and Urban Land (U. S. Portion) to The Great Lakes.

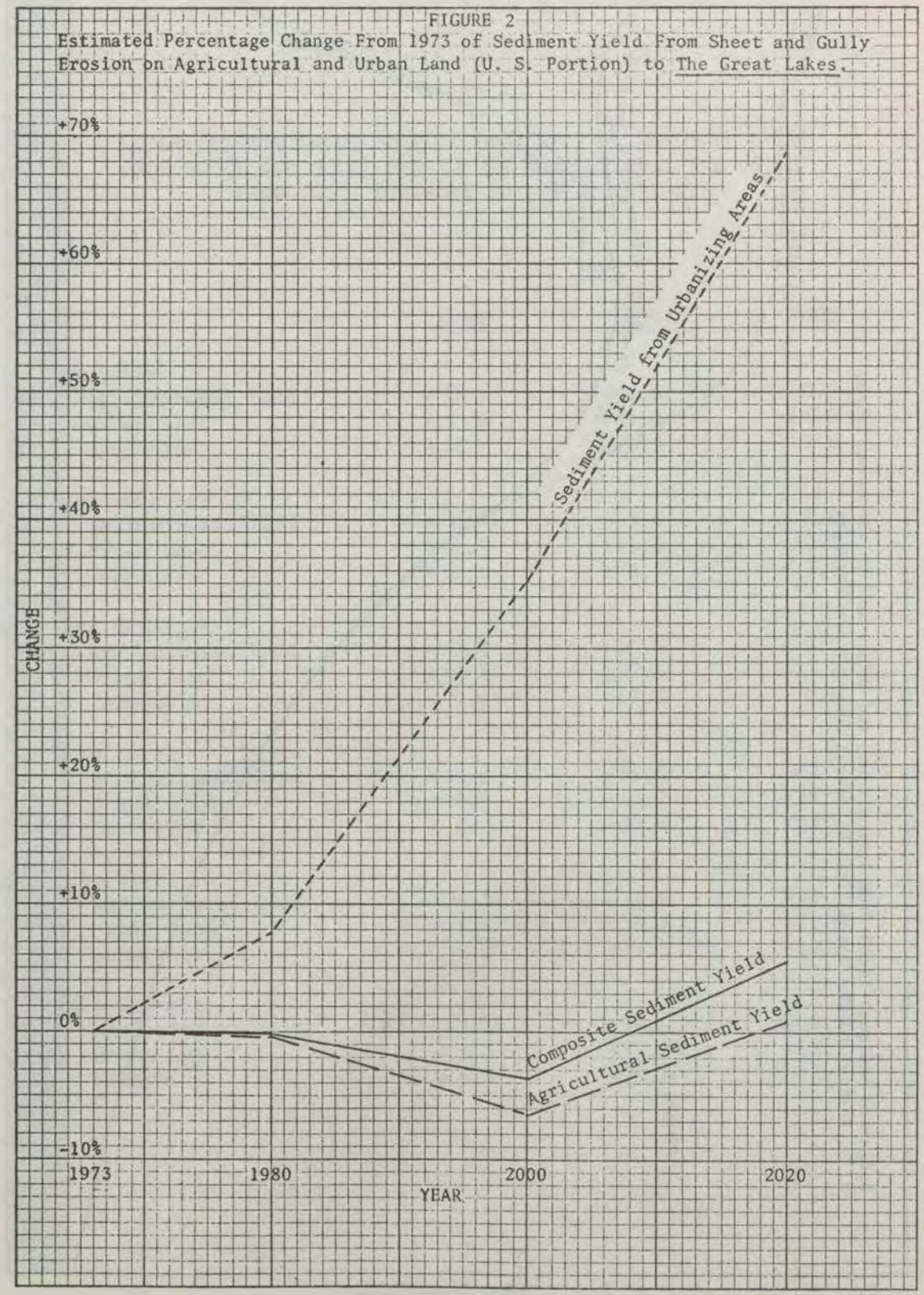


FIGURE 3

Estimated Percentage Change From 1973 of Sediment Yield From Sheet and Gully Erosion on Agricultural and Urban Land (U. S. Portion) to Lake Superior.

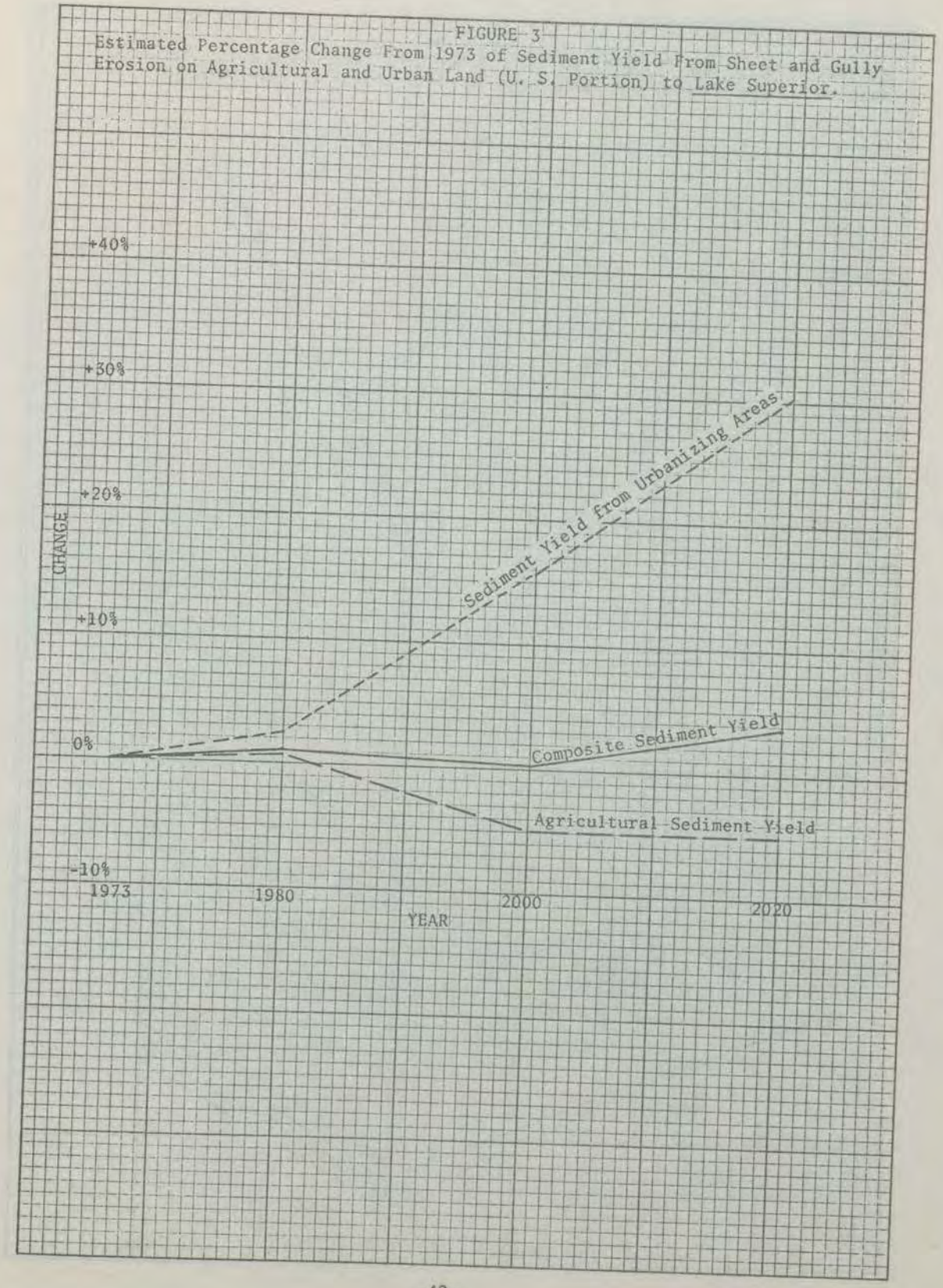


FIGURE 4

Estimated Percentage Change From 1973 of Sediment Yield From Sheet and Gully Erosion on Agricultural and Urban Land (U. S. Portion) to Lake Michigan.

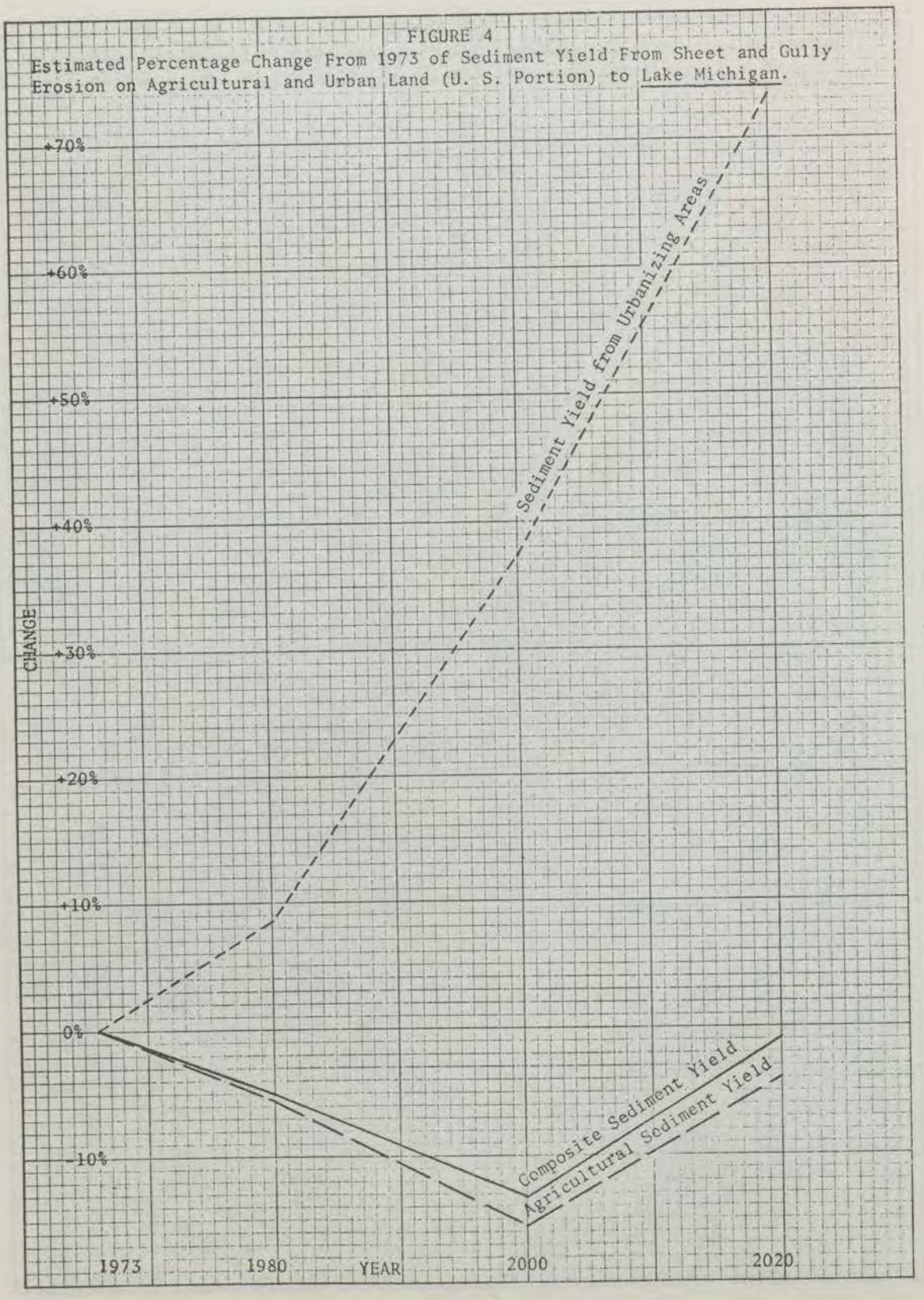


FIGURE 5

Estimated Percentage Change From 1973 of Sediment Yield From Sheet and Gully Erosion on Agricultural and Urban Land (U. S. Portion) to Lake Huron.

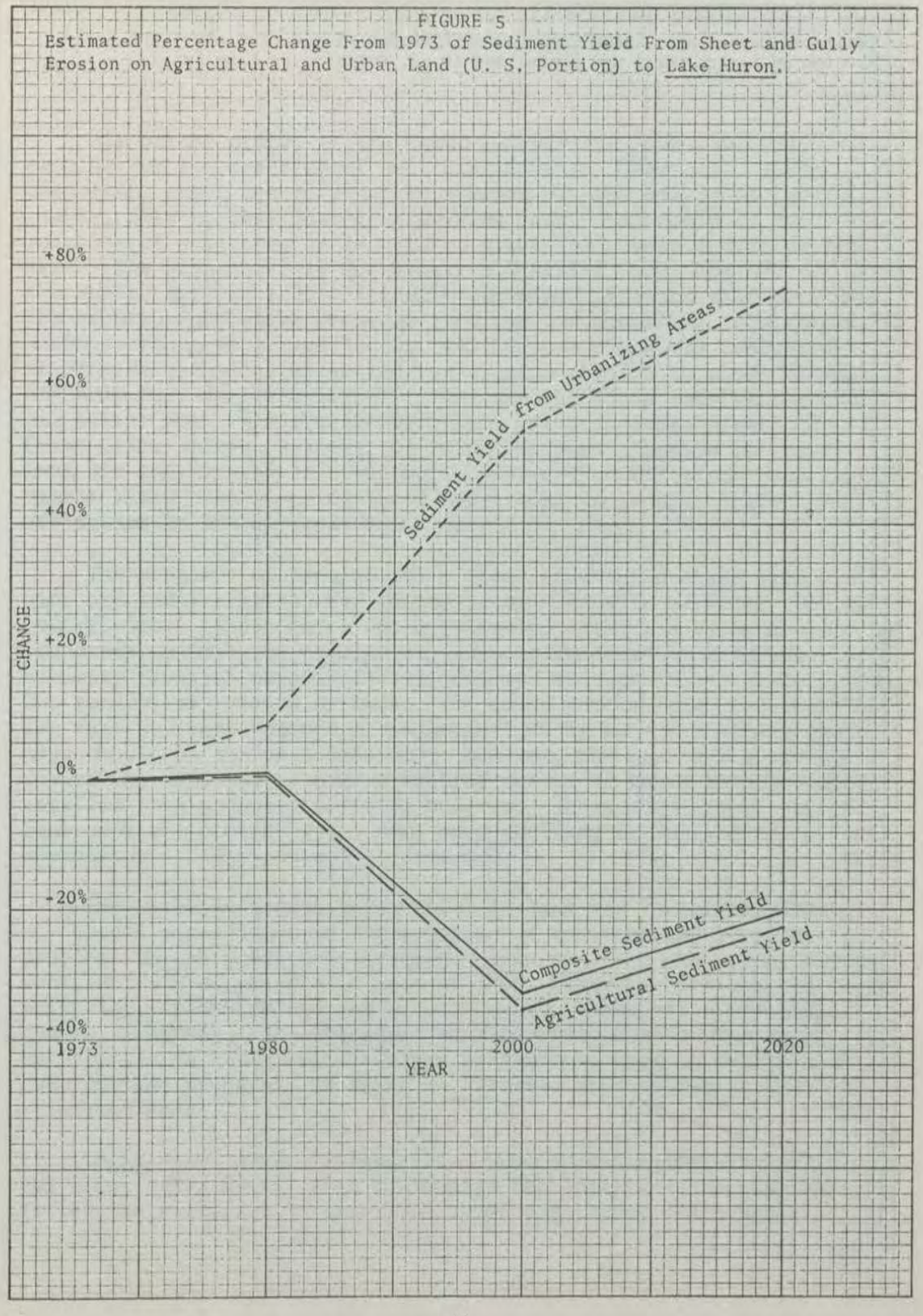
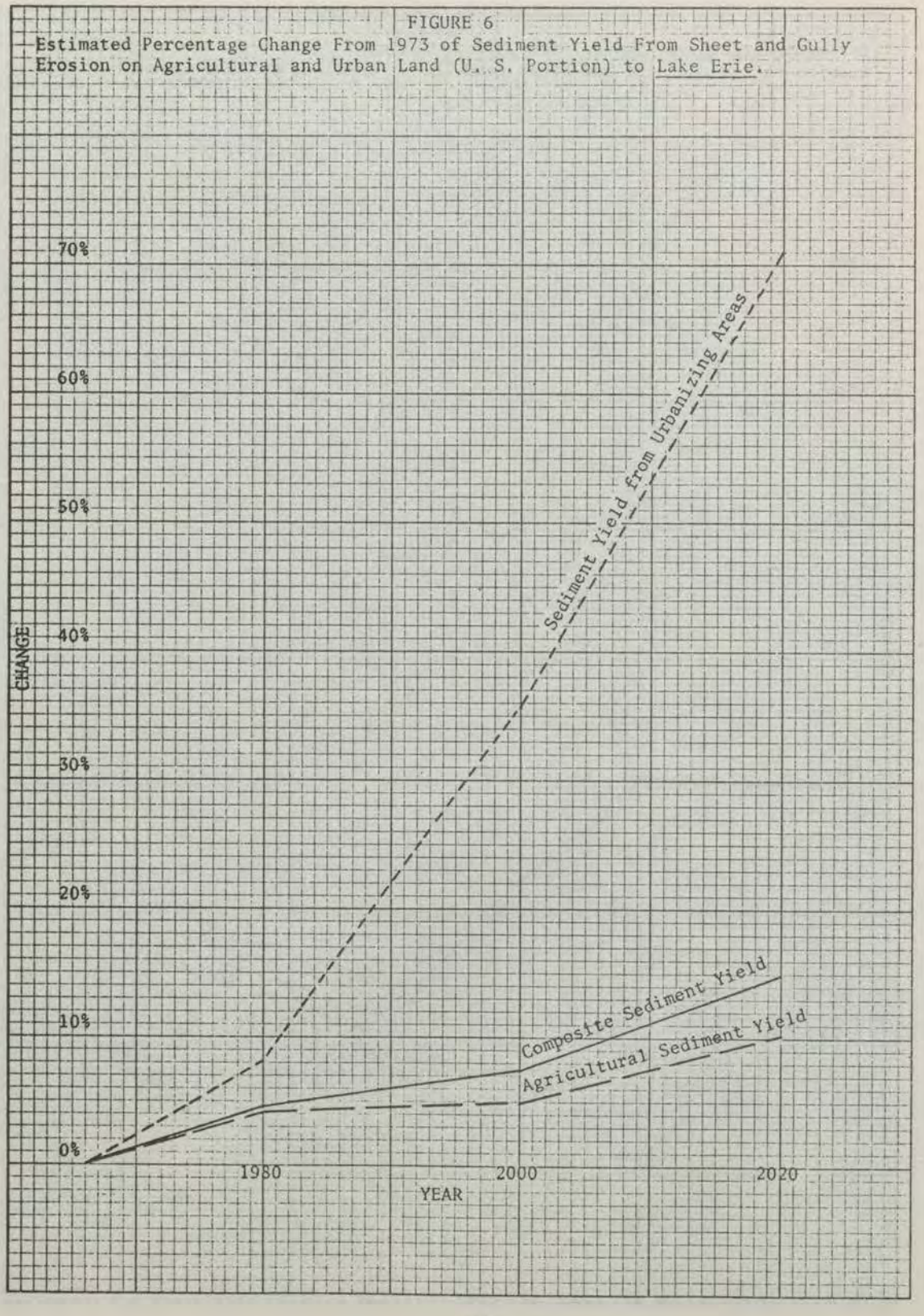


FIGURE 6

Estimated Percentage Change From 1973 of Sediment Yield From Sheet and Gully Erosion on Agricultural and Urban Land (U. S. Portion) to Lake Erie.



Section III-9

REVIEW OF INSTITUTIONAL ARRANGEMENTS TO REGULATE AND CONTROL SEDIMENTATION

All of the states in the Great Lakes Basin have laws that provide for the creation of soil conservation districts. These districts, which are governed by a board of supervisors of local people, fill a vital need in providing local leadership to the solution of soil erosion as well as other soil conservation problems. The Michigan Soil Conservation Districts Law, Act 297 of the Public Acts of 1937 originally contained sections that provided for adoption and enforcement of land use and treatment regulations. These sections were repealed in 1945 because districts preferred to provide technical assistance on erosion control to those requesting help on a voluntary basis. This general theme carries through all of the Soil Conservation District Acts activities within the Great Lakes States, a spirit of voluntary participation in sound land use and management. The soil conservation districts have in recent years been revising their operations to provide their assistance to non-agricultural areas.

The problems of detrimental effects to water quality from erosion and sedimentation have generally been handled by water quality control acts. These acts, which generally establish water quality standards, handle the problem of sediment in terms of allowable concentrations of suspended colloidal and settleable materials. The control of erosion by these acts would come through the reasoning that sediment becomes a "pollutant" as it enters a stream and thus should be abated. Generally, however, the laws are geared toward the handling of a much smaller number of cases than would be entailed in a sediment policing program. These acts, then, are generally not backed by the necessary staffing or funding to administer a sediment control program.

The Michigan Real Estate - Plat Act and its recent amendments may become significant in the solution of sediment and erosion problems in the Great Lakes region. This act, and its recent amendment, Act 288 of the Public Acts of 1967, State of Michigan, is primarily a planning and zoning act passed to encourage orderly growth. However, this act could provide authorization for local authorities to influence and bring pressure to bear on design and construction procedure in order to minimize erosion during the construction phase.

Legislative needs include state-wide enabling legislation in the Great Lakes States that would give the local governmental entities the authority to establish broad regulatory ordinances.

This authority should be given to existing entities of state and local government or to new community action entities. The implementation of these local programs should probably be done utilizing cut-off date procedures. By this method, local government would be given ample time to establish their own code of ordinances which would fit their particular situation. Beyond this date, if the local government had not established such regulation, state established regulations would prevail. The guidelines for these local ordinances should follow the commonly accepted standards and specifications for erosion control measures and the standards for water quality recommended by the state agencies should be followed.

Sediment control ordinances have emerged, to date, primarily in the rapidly developing urban areas as a means to give some measure of protection to local water quality and from other problems associated with sedimentation. Following is a sample ordinance, taken from the Community Action Guidebook for Soil Erosion and Sediment Control as an example. This Sedimentation Control Ordinance, for Montgomery County, Maryland, is one of the earlier regulatory measures of this type in the United States.

In the spring, 1965, a sediment control task force, formerly appointed by the Montgomery County Council, completed development of a proposed sedimentation control program for the county. The Council subsequently made the program stated county policy and solicited the voluntary cooperation of the building industry. Two years later (6/27/67), the council made sedimentation control mandatory through adoption of an amendment to the county's subdivision regulations. Chapter 104, as codified in the Montgomery County Code. This amendment, which represents the county's sedimentation control ordinance, follows:

Amend Sec. 104-24 Preliminary Subdivision Plan-Approval Procedure by adding new subsection (i) as follows:

(i) Sediment Control. The approval of all preliminary plans and extensions of previously approved plans shall include provisions for erosion and sediment control, in accordance with the Montgomery County Sediment Program, adopted by the County Council June 29, 1965.

(1) The Board, in its consideration of each preliminary plan or extension of previously approved plan shall condition its approval upon the execution by the subdivider of erosion and sediment control measures to be specified by the Board after receiving recommendations from the Montgomery Soil Conservation Sistrict.

(2) One copy of each approved preliminary plan or extension of previously approved plan shall be referred to the Montgomery Soil Conservation District for review and recommendation as to adequate erosion and sediment control measures to prevent damage to other properties.

(3) The installation and maintenance of the specified erosion and sediment control measures shall be accomplished in accordance with standards and specification on file with the Montgomery Soil Conservation District.

(4) Permits for clearing and grading prior to the recordation of plats shall be obtained from the Department of Public Works subject to the granting of temporary easements and other conditions deemed necessary by the Department in order to insepct and enforce the performance of the specified erosion and sediment control measures provided for in sub-section (1) above.

(5) In the event the subdivider proceeds to clear and grade prior to recording of plats without satisfying the conditions specified under Sec. 4, the Board may revoke the approval of the preliminary plan or extension of previously approved plan.

Amend Article 1, Section 23-2 General Requirements (of subdivision plans) by the addition of a new paragraph to be known as 23-2 (1) to read as follows:

(1) Erosion and Sediment Control Measures.
Adequate control of erosion and sedimentation of both a temporary and permanent nature shall be provided during all phases of clearing, grading and construction as approved by the Director.

Amend Section 23-8, Preliminary Plats - Preparation by the addition of a new paragraph to be known as 23-8 (g) to read as follows:

(g) Statement that Erosion and Sediment control methods shall be provided prior to any clearing, grading or construction

Amend Article 2 of Chapter 23 by the addition of a new paragraph to Section 23-12. Final Plats - Approval to be known as 23-12 (c) to read as follows: (Plats shall be approved only if)

(c) Plans and specifications for the control of erosion and sedimentation, if such controls are deemed necessary, have been submitted and approved by the Director of Public Works or his agent. This approval shall be concurrent with the approval of the aforesaid plans and specifications, and become a part thereof.

A number of states within the Great Lakes Basin have now developed ordinances and regulations on the control of erosion and sediment damage (9). Michigan's Soil Erosion and Sediment Control Act of 1972, approved by the Governor on January 9, 1972 provides for a statewide soil erosion and sedimentation control program with provision for inforcement by counties and for cooperation with soil conservation districts. Pennsylvania in 1972 amended a law to require approval by the Department of Environmental Resources on erosion and sediment control plans for earth moving activities except where an erosional and sediment control plan had been developed by the U. S. Soil Conservation Service. The law from both Michigan and Pennsylvania is unique in that it requires a permit before the farmer can cultivate his land unless he has a conservation plan approved by the local Soil Conservation District. Ohio, in 1971-1972 amended an act which would authorize the Director of Natural Resources to develop a procedure to coordinate agrucultural pollution abatement and urban sediment control programs based on the state's air and water quality standards. Illinois, in 1971, passed an act which requires soil conservation districts to furnish information to local zoning authorities and assist them in reviewing applications for proposed land subdivisions. Wisconsin, in 1971, amended its law to permit adoption of land use regulations by a simple majority of electors in lieu of two-thirds of the land occupiers, broadened its regulatory authority to include specifically the control of runoff and sedimentation, and provided for county assistance in enforcement of the regulations.

In addition several counties within the Basin have adopted ordinances on sediment control (9). Some of them are Otsego County, Michigan, 1971, Steele County, Minnesota, 1970; and Buffalo County, Wisconsin, 1965. Most of these ordinances require a permit for any earth moving activity, such permit to be issued only if the applicant files a plan for the control of erosion and sedimentation which is approved by the local Soil Conservation District. In addition, many cities, towns and townships within the respective states have erosion and sediment control legislation.

As an aid in drafting the above mentioned legislation, there has been available since September 1972 the Model State Act for Soil Erosion and Sediment Control suggested by the Council of State Governments (6).

The work group on Erosion and Sedimentation in the Great Lakes Basin Framework Study took the position that regulatory measures for the control of erosion and sedimentation should be adopted. The spirit of the regulation should be not only to eliminate "nuisance" type erosion situations but it should also be directed toward real protection, preservation, and enhancement of water and related resources.

Voluntary action to minimize detrimental effects from the inflow of sediment and other solids should be strongly encouraged. The community action guidebook referred to above recognizes that the sedimentation control programs that appear to work best are those that initially evolve from some type of voluntary action. The voluntary approach, as far as it can be effectively exploited to achieve real results, should be itilized.

The work group recommends a second avenue of approach which is parallel to the purely voluntary. This is the utilization of a system of incentives. Agriculture has had a program of this type for years in the form of the Agricultural Conservation Program (ACP). Under this program farmers were reimbursed for installing agricultural conservation practices - many of them for erosion control. This program, which was replaced by the Rural Environmental Assistance Program and which in turn has been succeeded by the Rural Environmental Conservation Program, will provide farm operators an incentive to apply erosion control measures. The Rural Environmental Conservation Program includes a new approach to erosion and sediment control on agricultural lands through the use of long-term (3 to 10 years) cost share contracts. These contracts have as their base a conservation plan which has been approved by the local soil conservation district. Permanent type land treatment systems are encouraged in the plan. The Rural Environmental Conservation does not now apply to the urban sector. An expanded incentive program could be developed to include the urban sector as it relates to runoff of sediment and other solids. The incentive could be in the form of cost-share on measures used to reduce erosion or a special tax write-off, justified on environmental enhancement.

The work group recognized the fact that control ordinances are essential in order to achieve the results desired by society in many situations. The problem of sedimentation is one of these. Ordinances designed to control the inflow of sediment from erosion and other sources may be essential in order to approach the levels essential for real water quality.

Regulation to control sedimentation and suspended solids must be uniformly applied to all sources if real water quality standards are to be effected. The sections above point out the many diverse sources of solids both by erosion and from other waste. Thus if an ordinance is established to regulate the subdivision developer because of "pollution" resulting from erosion

and sedimentation, then why should not the farm operator be regulated the same way? Both types of operations, under lax management, can produce sediment that becomes a nuisance and affects local water resources in the same way. But it does not stop here. There is ample evidence that waste from established urban areas contributes large quantities of solids to the adjacent water courses, lakes, etc. These solids constitute the same debasement of water quality as do solids from erosion of land. Thus when we point the finger at a "sediment polluter" we end up with a round-robin situation. The urban people point to the sub-division developer, the developer points to the farmer, and the farmer points back to the urban areas.

Section IV

SUMMARY OF RELEVANT ONGOING OR PROPOSED RESEARCH AND DEMONSTRATION PROGRAMS

The following studies are being conducted in or in close proximity to the Great Lakes Region and are thought to be relevant to this report.

Income Effects of Alternative Farm Plans which Achieve Soil Conservation.

Objective - Given soil losses specified as acceptable by authorities, income effects of variation of following factors will be estimated for typical farm situations in selected problem areas with relatively uniform soil conditions: capital availability; length of period which costs and returns are discounted; substitution of mechanical erosion - control practices for vegetative practices; minimum tillage will also be considered, and leasing system. Estimate effect of soil losses on income above recommended levels.

Investigator - E. R. Swanson, University of Illinois, Urbana, Illinois.

Sponsor - University of Illinois, Agricultural Experiment Station, Urbana, Illinois.

Time frame of studies - 1967-1973.

New information likely to be obtained - The effect on income of utilizing soil conservation practices which reduce erosion and sediment yield.

Chemical and Sediment Movement from Agricultural Land into Lake Erie.

Objective - Determine the quantity and kinds of soluble and suspended constituents in water which drain into Lake Erie. Evaluate the effect of different drainage methods, tillage operations, and rainfall characteristics on constituents entering Lake Erie from agricultural lands. Determine the rates of leaching of phosphorus from fine and coarse textured soils, identify discute compounds formed, and determine threshold concentrations where soil contributes phosphorus to lake water.

Investigator - G. O. Schwab, E. D. McLean, A. C. Waldron and R. K. White, Ohio State University, Columbus, Ohio.

Sponsor - Ohio State University Water Resources Center, Columbus, Ohio.

Time frame of studies - July 1969-Sept 1973.

New information likely to be obtained - Effect of tillage practices on sediment and drainage water quality.

Pollution and Movement by Activation Analysis Principles.

Objective - Determine the sources of sediment contributing to the pollution of streams and rivers. Determine the pattern of sediment movement in a field to check theoretical mechanics of erosion equations. Determine the change in total sediment load per unit of area as watershed size increases. Relate sediment production to storm characteristics.

Investigator - C. E. Beer and H. P. Johnson, Iowa State University, Ames, Iowa.

Sponsor - Iowa State University, Agricultural Experiment Station, Ames, Iowa.

Time frame of studies - 1969-1974.

New information likely to be obtained - Sediment delivery ratio for small watersheds.

Contribution of Soil Detachment and Transportation by Raindrop Action to the Total Erosion from an Area.

Objective - Determine relative significance of raindrop erosion as opposed to runoff erosion. Evaluate erosion process in a rill and between rills to determine where infiltration occurs.

Investigator - R. A. Young, Agricultural Research Service, Morris, Minnesota.

Time frame of studies - 1969-1975.

Sponsor - Agricultural Research Service.

New information likely to be obtained - The determination of dominance of raindrop erosion or runoff erosion.

Field Determination of Nutrient and Sediment from Non-point Sources.

Objectives - Relate water quality and sediment yield to land use practices. Provide data for an agricultural chemical transport model.

Investigator - R. A. Young and C. A. Onstead, Agricultural Research Service, Morris, Minnesota.

Sponsor - USDA, Agricultural Research Service.

Time frame of studies - January 1973-January 1974.

New information likely to be obtained - Refine the effect of land use practices on sediment yield.

Mechanics and Control of Soil Erosion by Water.

Objective - Define principles involved in soil detachment and transport by rainfall and water movement across or through the soil. Develop and test practices that use crop residues, tillage techniques and soil treatment to reduce erosion.

Investigator - W. H. Wischmeier, USDA, Agricultural Research Service, Lafayette, Indiana.

Time frame of studies - 1969-1974.

Sponsor - USDA, Agricultural Research Service, Lafayette, Indiana.

New information likely to be obtained - Data which can be used to improve predictive equations.

Black Creek Study - Program for Reduction of Sediment and Related Pollutants in the Maumee River and Lake Erie.

Objective - Determine the characteristics which would be necessary to conduct a meaningful demonstration and research project on a small watershed. An investigation of the Black Creek Area identified land treatment measures, which will significantly reduce the sediment contribution from this watershed to the Maumee River. Monitoring sites were selected within the watershed and a plan of investigation which will lead to a projection of results of the demonstration project to the basin was developed. Also developed were a series of scientific studies to aid in the understanding of the mechanisms involved in the treatment of the watershed. A work schedule for treatment was developed and specific areas of concern identified.

Investigator - Allen County Soil and Water Conservation District, Soil Conservation Service, Agricultural Research Service, and Purdue University.

Sponsor - Environmental Protection Agency.

Time frame of studies - October 1972-October 1977.

New information likely to be obtained - Effect of land treatment on water quality in streams.

Pilot Erosion and Sedimentation Monitoring and Accelerated Land Treatment Project.

Objective - Measurement of effects of land treatment on erosion and water quality.

Investigators - Michigan State Department of Agriculture, Hillsdale, Michigan.

Time frame of studies - 1974-1979.

New information likely to be obtained - Magnitude of erosion and sediment yield from this area.

Kalamazoo - Blacks - Paw Paw Type 4 Study.

Objective - Analysis of land use planning, erosion control and environmental protection.

Investigators - Soil Conservation Service, Economic Research Service, Forest Service and Michigan Department of Natural Resources, East Lansing, Michigan.

Time frame of study - 1973-1976.

Sponsor - Michigan State Department of Natural Resources.

New information likely to be obtained - Effect of land use and management and erosion control on the environment.

Grand River Basin Type 2 Study

Objective - A broad water resources study to determine flood control, land management and recreation problems and provide proposals for their solution.

Investigators - All concerned federal and state agencies most of which are located at Lansing, Michigna.

Time frame of studies - 1962-1973.

Southeast Michigan Type 4 Study

Objective - Analysis of flooding, drainage and land use and management of the area.

Investigators - Soil Conservation Service, Economic Research Service and Forest Service, East Lansing, Michigan.

Time frame of studies - 1967-1974.

Sponsor - Corps of Engineers and State of Michigan, Department of Natural Resources, East Lansing, Michigan.

New information likely to be obtained - An analysis of how land use, flood control, drainage, land treatment, and erosion control effect each other.

Prediction of Erosion and Sediment Transport from Agricultural Watersheds.

Objective - To model rainfall-runoff-erosion and transport processes on a cropped watershed. To determine amounts of erosion from small single crop watersheds as affected by soil type, land use treatment, and rainfall characteristics.

Investigators - D. C. Kincaid and W. M. Edwards, Coshocton, Ohio.

Sponsor - USDA, Agricultural Research Service.

New information likely to be developed - Modeling work will result in a greater understanding of the process of detachment, transport and deposition of sediment on a watershed. This knowledge will aid in the prediction of chemical transport on sediment. The erosion data from small watersheds will be analyzed with a modified Universal Soil Loss Equation that includes runoff as an erosion and transport vehicle to obtain improved estimates of the crop management and erosion control practice factors.

Erosion and Sediment Inventory in New York State.

Objective - Determination of the gross erosion and resulting sedimentation occurring in New York.

Investigator - USDA, Soil Conservation Service, Syracuse, New York.

Time frame of study - 1971-1974.

Sponsor - USDA, Soil Conservation Service in cooperation with Soil and Water Conservation Districts.

New information likely to be obtained - The magnitude, location and extent of erosion and sedimentation problems in New York State.

Section V

NATURE AND AVAILABILITY OF TECHNOLOGY TO COPE WITH POLLUTION PROBLEMS IDENTIFIED

The best measures to deal with sedimentation problems on agricultural and urban lands are known as soil conservation practices. These practices, based on research and practical use, have been developed over a long period of time and are known to be effective when properly applied. A land treatment system is made up of a combination of conservation practices needed to do the control job. Each practice in the system has a specific and often critical relationship to the other practices. For example, minimum tillage can and does reduce erosion but the erosion is reduced to a greater degree when this practice is used in conjunction with other applicable practices. The total land treatment system is more effective in reducing erosion than the sum of its component parts.

The following list of practices is by no means complete, but it covers the most important ones in use in the Great Lakes Basin.

Conservation Cropping Systems - Growing crops in combination with needed cultural and management measures particularly the use of close growing crops on erodible soil.

Contour Farming - Conducting farming operations on sloping land in such a way that plowing, land preparation, planting and cultivating are done on the contour.

Cover and Green Manure Crop - A crop of close-growing grasses, legumes, or small grain used primarily for seasonal protection and for soil improvement.

Critical Area Planting - Stabilizing severely eroded areas by establishing vegetative cover.

Crop Residue Management - Utilizing and managing crop residues for soil protection on a year round basis or when critical erosion periods usually occur.

Diversion - Channel constructed across the slope for the purpose of intercepting runoff.

Grade Stabilization Structure - A structure to stabilize the grade or to control head cutting in natural or artificial channels.

Grassed Waterway or Outlet - A natural or constructed waterway or outlet shaped or graded and established in suitable vegetation as needed for the safe disposal of runoff.

Minimum Tillage - Limiting the number of cultural operations to those that are properly timed and essential to produce a crop and prevent soil damage.

Mulching - Applying plant residues or other suitable materials not produced at the site to the soil surface.

Pasture and Hayland Management - Proper treatment and use of pasture or hayland.

Proper Grazing Use - Grazing at an intensity which will maintain enough cover to protect the soil and maintain or improve the quantity and quality of desired vegetation.

Sediment Debris Basins - Storage for sediment provided by a dam with spillway above channel grade; by excavation below grade or both. Water retention is not an intended function of the structure.

Stream Channel Stabilization - Stabilizing the channel of a stream with suitable structures.

Stripcropping - Growing crops in a systematic arrangement of strips or bands across the general slope or on the contour to reduce water erosion.

Terracing - Development of water storage capacity along the contour by excavation and placement of soil as an embankment along the downstream side. Intervals vary with precipitation, soil and slope.

Urban land is treated with many of these same measures. In fact the reasoning behind the following list of principles for sediment control on urban land follows the same reasoning used for agricultural land.

1. Expose the smallest practical area of land at any one time during construction. Clear only street rights-of-way initially; do not disturb the remaining areas until storm drainage systems and streets, curbs and gutters are installed.

2. Expose the area for the shortest practical period of time. Schedule construction so that only a limited amount of land is exposed during periods of high intensity storms which are most likely to cause erosion.

3. Make temporary plantings or install land treatment measures to protect critical exposed areas. Stabilize those graded lots which will remain exposed for an appreciable period of time with temporary vegetation. Provide temporary diversions to intercept water flowing down slopes and convey that water to a non-erodible outlet.

4. Install and maintain sediment traps or basins to separate the sediment from runoff water.

5. Accommodate increased runoff. The changed soil and surface conditions, particularly the addition of impervious areas, increases the volume of runoff. Provide for storm water management which includes onsite storage.

6. Establish permanent vegetation and structures as soon as possible.

7. Fit the development plan to topography and soils so that erosion potential is lowest. Alter the development to fit the ground, not the ground to fit the development. This reduces the amount of grading required and allows the retention of the maximum amount of natural vegetation.

8. Where feasible, natural vegetation should be retained and protected. Plan homesites to make optimum use of existing topography and vegetation.

It can readily be seen that many of the soil conservation practices listed previously are useable or can be adopted with slight modification for urban use.

These soil conservation practices when properly applied are very effective in reducing erosion. The problem is to get all the necessary practices applied. It is estimated that approximately 64% of all agricultural land in the U. S. portion of the Basin still needs some treatment (4). The present erosion rates are about 80% of what they would be if none of the soil conservation practices had been established in the Basin (5). If all soil conservation practices needed for erosion control were established, erosion rates would approach "natural geologic erosion rates." Unfortunately soil conservation programs like all human endeavors, are in practice seldom 100% effective.

Restrictions which limit further reduction of erosion and sediment yield rates by the use of soil conservation practices are mostly economic. A farmer would rather plant a cash crop than one designed just for erosion control as he gets a greater immediate return from the cash crop. Efforts to stimulate motivation, interest, and acceptance of new ideas is needed.

If a massive program to achieve conservation land use and treatment was started, it would not be completely effective because of various economic and personal constraints. Therefore, considering the soil conservation program as it is apt to be applied the reduction in erosion and sediment yield might be 70% effective (5).

When compared to other Basins in the U. S., the sediment problem is not as great. Of course local problem areas exist and it is important to eliminate these. The most important reason for reducing the pollution caused by sediment from agricultural and urban land is to minimize the amount of other pollutants coming to the Lakes. Sediment is the principal means by which other pollutants are carried to the Lakes.

The technology in the field of soil conservation is well enough developed so that erosion can be held within tolerable limits if the proper practices are installed. Frequently, however, there has to be a trade off against economics. In these cases either a less desirable or effective soil conservation practice is installed or nothing is done. It is in this area where there is a need for new technology. For instance, in some portions of the Basin it is the practice of the farmers to plow the land in the fall for a spring planting. Fall plowing leaves the land unprotected from precipitation and wind and in the worst possible condition as far as erosion protection in winter and spring is concerned. The farmers say they fall plow because winter precipitation leaves the ground too wet to plow in the spring and without plowing they cannot plant and grow a crop. Techniques for protection of these types of lands need to be improved.

Section VI

NEED FOR NEW RESEARCH, DEMONSTRATION OR MONITORING PROGRAMS

As pointed out in section III-4 techniques for predicting sediment yield from large watersheds or from urban land are not very precise. In order to make more accurate predictions on sediment yield to the Lakes many more sediment sampling stations on streams need to be established. These stations should be located close to the mouth of the stream so as to measure as much of the watershed as possible. Also, additional sediment sampling stations should be located within the watershed to help refine source information. If stations in the upper watershed were monitored and the data correlated with changes in land use considerable useful information could be acquired on the effect of land use and treatment on sediment yield. Installation of these stations and acquisition of data from them might be considered research, data collection, or a monitoring program. In order to make the predictions which are based on this data as accurate as possible the period of record of the stations should be as long as necessary to obtain suitable information. A minimum period of record is suggested as 3-5 years but some stations might be needed for 10 years or longer.

Since it is not likely that all streams flowing into the Lakes can be measured and sampled it would be desirable to have other methods of predicting sediment yield. One method is by use of gross erosion and sediment delivery ratios. There are reservations about the validity of the sediment delivery ratio curve on larger watersheds. Research on sediment delivery ratio is needed to refine this method.

Comparison of a measured sediment accumulation from a watershed to another under study is one more technique for predicting sediment yield. In order to use this method with confidence many more reservoir sediment surveys must be made. These surveys should be on reservoirs which have a variety of watershed sizes. Once measured selected reservoirs should be measured again on an interval determined by conditions in the watershed. If a watershed changes from agricultural to urban land use sediment surveys in a downstream reservoir can furnish sediment yield rates for agricultural, urban construction, and urban land use. Measuring sediment in reservoirs could be considered research, long ranged monitoring programs and data gathering.

Where there is no need for gross erosion information and all that is desired is an estimate of the sediment yield at a given point predictive equations to furnish this information can

be very useful. To date no equations of this type have been developed for this area. Basic research to develop an equation of this type would be very desirable.

Research, data collection and long ranged monitoring programs could be used to acquire information on sediment from urban construction activity and settled urban areas. The magnitude of the factors effecting sediment yield from these areas have not yet been defined.

Some research is needed to develop techniques to protect those lands where the farmers practice fall plowing. The research could be in the form of proof that fall plowing was not needed with the present level of technology or in the development of agronomic and mechanical practices which would furnish protection to the land during the critical winter period.

Section VII-1

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Land Use/Water Quality Relationships in the U. S. Great Lakes Basin
Task A: To assess problems, management of programs and research...
Combined Categories: A8 - Animal Wastes
A9 - Intensive Animal Feedlots

WATER POLLUTION PROBLEMS ASSOCIATED WITH ANIMAL WASTES AND INTENSIVE ANIMAL FEEDLOTS IN THE UNITED STATES PORTION OF THE GREAT LAKES BASIN

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To be used as portion of the U. S. Task A Report on GREAT LAKES POLLUTION FROM LAND USE ACTIVITIES by the International Joint Commission - prepared in partial fulfillment of U. S. Environmental Protection Agency Contract No. 68-01-1598

March 1974

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Section 1

SUMMARY

1.1 Introduction

This report attempts to indicate the present and potential future problems associated with animal waste and intensive animal feedlots on land tributary to the Great Lakes. Attention is focused on surface and groundwater problems.

Relatively little land in the Basin is used for livestock production. The major land use is predominantly cropland and woodland. Land associated with dispersed livestock operations, pasture land, is estimated to be no more than 5-14% of the land in the Basin. The relative pollution potential from livestock operations is not necessarily in the same proportion as the land use since the pollutants per unit of runoff from feedlots and similar operations are higher than those from other land uses.

All types of livestock are produced in the Basin with dairying being the major livestock operation. In 1969, there were approximately 130,000 farms that had an income of $2500 per year or greater producing livestock in the Great Lakes Basin. Additional smaller farms also existed.

Few of the livestock operations are extremely large, there being only about 190 beef cattle feedlots with a capacity of 1000 head or greater in all of the states adjacent to the Great Lakes. Information is not available on the number of feedlots on land tributary to the Great Lakes.

The potential pollution problems from these livestock operations are contaminants in runoff from confined operations, from land used for manure disposal, and from pasture land. The potential contaminants are diverse including organics, inorganics, nutrients, bacteria, solids and soluble material.

1.2 Summary of Findings and Conclusions

The water pollution problems associated with animal wastes and intensive animal feedlots in the Great Lakes Basin are those due to runoff from land used for animal

production. The definition of an intensive animal feedlot is arbitrary. If one uses the definition of a feedlot that appears in the EPA regulations for the feedlot point source category, i.e., operations having 1000 head of beef cattle, 700 head of dairy cattle, 2500 swine, 10,000 sheep, 55,000 turkeys, 5,000 ducks, or poultry operations with a liquid system, there are very few intensive livestock operations in the Basin.

However, there are large livestock operations in the Basin. By using a definition of an intensive, single specie operation as an operation having either 100 or more head of cattle, 200 or more hogs and pigs, or 10,000 or more poultry, the location of the large operations were identified by counties. The major livestock production areas are southern Wisconsin, southern Michigan, western Ohio, and northern New York. Streams draining these areas will be the main transporters of any potential runoff pollution from livestock operations to the Great Lakes.

The livestock operations in the Great Lakes Basin are less intensive operations than the large beef cattle feedlots in the midwest that have attracted national attention due to size and water pollution potential. Average livestock population per farm in the states adjacent to the Great Lakes ranged from 44 to 62 cattle and calves per farm, 28 to 157 hogs and pigs per farm, and 340 to 3220 poultry per farm. These operations commonly are livestock-crop production operations with available land for waste disposal as well as the need for the fertilizer value of the wastes. The actual water pollution problems in the Basin due to animal wastes are largely unknown and can only be suspected.

Technology and guidelines are available to control the point source runoff from intensive livestock operations and to minimize the contamination from land used for manure disposal and from pasture land. The EPA feedlot point source regulations that became effective on April 15, 1974 appear adequate to control the pollution from the large livestock operations as well as others who have or will cause pollution problems. Assuming that the EPA regulations will be enforced, point source animal waste pollution problems will be less severe in the future. Non-point pollution problems probably will be the same or slightly less in the future.

There appears to be no need for state pollution control activities to be more stringent than the federal activities with regard to wastes from animal production operations. One of the largest concerns is how and if the regulations will be enforced. The many diverse livestock operations will require flexible interpretation of the regulations and knowledgeable individuals to achieve the pollution control objectives. The design and regulatory personnel that are needed can be a constraint on the rapid implementation of animal wastes pollution control in the Basin.

Very little information is available to assess the groundwater pollution problem associated with livestock operations in the Basin. Such problems would be localized problems if they do exist. Greater information would be useful in this area.

A considerable amount of research has been conducted and is underway to find technical solutions to animal waste management problems. However, there are few apparent projects that relate surface and groundwater quality to livestock density, land waste application rates, or other important management variables. Few existing projects appear broad enough to permit unit quantitative values or predictive cause and effect relationships to be developed. Of particular need are studies that will identify the changes in pollutants that take place as contaminated runoff is transported across land and in streams.

1.3 Recommendations

Greater details on these recommendations can be found in Section 9. Based upon the information gathered for this report, it is recommended that:

a - A groundwater monitoring program should be established to assess the changes in groundwater in and around livestock operations, including confined operations, pasture land, and land used for manure disposal.

b - Several comprehensive surface water monitoring programs should be established to relate runoff characteristics to land and waste management conditions and to develop predictive quantitative pollutant loss relationships

c - Detailed studies should be instituted to quantify the changes that take place in runoff contaminants as the runoff is transported across land and in streams. Such studies should be broad enough to identify changes in chemical, physical, and bacteriological parameters as well as to take annual variations in hydrologic events into account.

d - Field scale studies should be instituted to establish land application rates for animal wastes that will minimize surface and groundwater pollution problems.

e - An analysis should be made to determine how the federal and any state regulations will be enforced to control animal waste pollution problems in the Basin, if adequately trained personnel are available to assist producers and to staff pollution control agencies, and the level of personnel and financial support that is necessary, if any, to accomplish such pollution control.

f - Each state or region should be encouraged to develop good practice guidelines to help the livestock producer understand the magnitude of the pollution problems that may result from animal production operations, the regulations he faces, the technology he can apply, and the assistance he can obtain.

Section 2

INTRODUCTION

2.1 General

Historically, major efforts to maintain and enhance the water quality of the nation have focused upon problems caused by urban centers. This emphasis has been due to pressing problems in controlling industrial pollution, in treating domestic liquid wastes, in disposing of municipal solid wastes, and perhaps because of an instinctive feeling that agriculturally related environmental quality problems were uncontrollable and/or minor. In recent years, however, increasing attention has been given to the waste problems of agriculture.

The specific role of agriculture as it affects the water quality of the nation is unclear. The contribution of agriculture to water, air, and nuisance problems, in comparison to contributions from industrial and domestic sources, is difficult to assess. The available information suggests that the contribution from agriculture may be significant at the regional or local level. Data on fish kills from feedlot runoff, nutrient problems due to runoff from cultivated lands, the possible contamination of groundwaters from crop production and land disposal of wastes, and the increasing size of agricultural production operations indicate that the environmental consequences of these changes must be properly assessed in terms of regional, national, and international water quality needs and goals.

This report attempts to indicate the present and possible future problems and solutions associated with animal wastes and intensive animal feedlots on land tributary to the boundary waters of the Great Lakes. Specific emphasis is on surface and groundwater problems.

2.2 Scope of Study

The significance of pollution from livestock production operations has been the subject of many conferences, symposia, and detailed reports. Many papers have described the general and specific aspects of livestock wastes as they may affect water quality and cause water pollution problems. A list of the reports, proceedings, and books that are perti-

nent to the control of pollution from livestock wastes is noted in Section 10 - References.

Although this study focuses on the livestock operations and companion water quality problems in the Great Lakes Basin, the study was not confined to those reports and documents describing operations and problems in the Basin. Studies and data from all parts of the nation are able to provide insight to the magnitude and quantitative aspects of water pollution that may result from livestock operations. Relevant information available from the published literature, governmental reports, federal abstracting organizations, and conference proceedings were reviewed in developing this report. Studies that were particularly important to this assessment are cited in the subsequent sections.

The environmental problems associated with livestock production are relatively recent, being obvious to those most close to the problem for about 15 years and obvious to to the general public for no more than 10 years. Considerable effort in the early years was spent defining the magnitude of the problem and assessing the most suitable approaches for solutions to air and water pollution problems. Viable waste management approaches began emerging in the last five years. Current research and demonstration activities continue to clarify the technical and economic feasibility of various waste management alternatives.

Thus the time frame considered for this report covers the past 10 years. Most of the studies cited are more recent with some information becoming available as recently as late 1973 and early 1974.

2.3 Study Procedure

The work was accomplished by a detailed review of the large volume of pertinent material collected by the author over the past decade, by obtaining recent publications and reports from governmental agencies and available technical journals, and by phone and in-person discussions with individuals knowledgeable in many aspects pertinent to the study. Individuals in the Agricultural Research Service, U.S. Department of Agriculture and the Environmental Protection Agency (EPA) were contacted to obtain details of their most recent research activities related to livestock waste management. The Effluent Guidelines Division of EPA was contacted for the most recent version of the guidelines

pertaining to the Feedlot point source category. Individuals in the Economic Research Service, USDA, were contacted for the most recent information on the size of livestock operations. The 1969 U.S. Census of Agriculture was used to identify the size and number of livestock operations in the counties tributary to the Great Lakes. Individuals in several states were contacted for information on state regulations pertaining to management of livestock wastes.

The Great Lakes Basin Commission furnished economic and demographic information concerning agriculture in the Great Lakes Basin as well as a recent print out of abstracts available from the Current Research Information System (CRIS), USDA. In addition to the CRIS abstracts, notice of research projects from the Smithsonian Science Information Exchange were obtained and reviewed as were abstracts of articles on agricultural runoff from the Water Resources Scientific Information Center. Pertinent information from Canada and England also was reviewed.

All of this information was synthesized and incorporated into the report as appropriate. Specific information and references are cited where used. The few assumptions that were made are noted as they are first discussed.

2.4 National Overview

2.4.1 General

Agricultural production practices have not remained static in recent decades but have been in dynamic change to increase efficiency of production and to meet the demand for increased food supplies. All agricultural production is becoming more intensive. Farm size and productivity per farm worker have increased. It is unlikely that the ultimate in agricultural productivity has been reached and further increases in agricultural efficiency and production will be achieved. These changes have the potential to adversely affect surface and groundwater quality unless continued efforts are made to accompany these changes with improved waste management methods.

Population growth and consumer desire determine demand for agricultural products. As the standard of living within a county improves, a greater demand for livestock and livestock products occurrs and there is a decline in the

consumption of starchy foods. In the United States, the per capita consumption of beef and broilers has increased while that of dairy products, eggs, cereal products, pork and fresh fruits and vegetables has decreased. Since 1950, U.S. per capita consumption of all red meat, beef, and broilers has increased at about 0.8, 0.9, and 0.45 kg (1.8, 2.0, and 1.0 pounds) per year respectively. The increase in red meat consumption is due to the strong upward trend in beef consumption which has more than offset the decline in consumption of pork, lamb, and veal. In 1970, about 180% more chicken meat and about 240% more turkey meat were produced than in 1950. Chicken and turkey consumption per person increased about 110% during this period. Total U.S. per capita food consumption in 1970 was 6% greater than that in 1960.

If these trends continue, significant increases in beef cattle and broiler production will be needed to meet both the population growth and the increased per capita consumption. The production of other animal products will increase at a rate close to or slightly less than that of the population growth.

2.4.2 Livestock

To be economically competitive, livestock production has become concentrated in larger operations, become specialized in certain geographical areas, and used long distance transportation of inputs and outputs. The result has been increased confinement feeding of livestock and increased numbers of animals per livestock operation. Mechanization, improved production methods, and better nutrition and disease control have made it possible for the livestock producers to handle more animals with a minimum personnel increase. The scarcity of inexpensive farm help has supported the trend.

The most dramatic increase in livestock production has occurred in the broiler industry where the numbers of broilers raised in the United States increased from 1.8 billion in 1960 to almost 3 billion in 1970. During the same period, the number of beef cattle increased about 25 million head, an increase of about 38%.

Over two-thirds of the U.S. milk production is in the North Atlantic (19%), East North Central (28%), and West North Central (21%) states. The East North Central region consists of states tributary to the Great Lakes, i.e., Wisconsin, Illinois, Indiana, Michigan, and Ohio. Of the

other states tributary to the Great Lakes, New York and Pennsylvania are in the North Atlantic region and Minnesota is in the West North Central region. The three top milk producing states are Wisconsin, Minnesota, and New York.

Fewer but larger dairy farms will exist in the future as dairymen improve techniques to produce livestock products more efficiently and profitably. There are differences in milk production per cow between and within the regions. In the next decade, these variations will continue. Overall, the total milk production in the United States should remain constant or slightly decrease if milk production per cow increases at about the same rate the number of cows decrease. If there is a decrease in the energy content of dairy cattle feed or if numbers of milk cattle decrease because of increased meat demand, the total milk production will decrease more rapidly.

The demise of small, less than 30 cow dairy farms in Michigan has been predicted by 1980. Dairy farms with 30-50 cows are expected to remain constant in numbers while the farms having greater than 50 cows are expected to increase from 685 in 1959 to 2400 in 1980 and to contribute over 55 percent of the milk production in that state (1). In New York, it has been estimated that there will be 10,000 dairy farms in 1985, a decrease of 14,000 farms from those that existed in 1968. The average size of the herds in 1985 will be 80 cows, approximately double the herd size in 1968. The number of herds with 100 or more cows is estimated to triple by 1985 (2). Between 1960 and 1970, New York dairy farms with less than 30 cows declined by more than 70%. Farms with 50 or more cows increased by over 30% and farms with over 100 cows increased by about 110% during this period. Similar changes are expected in other states bordering the Great Lakes.

The changes in both the numbers and sizes of beef cattle feedlot operations have been dramatic. Specialization has removed many cattle from pasture and grass land and has resulted in confinement of large numbers in small areas with an average density of perhaps one animal per 4.5 to 14 square meter (50 to 150 square feet). During the 1960-1970 period, the number of cattle on feed, i.e., in feedlots, in the United States increased at the rate of over one-half million head per year.

Since mid-century, commercial feedlot operations, in which feed and water are brought to the animals in confined areas, have been expanding rapidly. Beef cattle enter such feedlots weighting 270-360 kilograms (600-800 pounds), are

fed a ration high in grain and protein concentrate, and gain about 0.9-1.4 kilograms (2-3 pounds) per day. The increase in commercial confined feedlot operations has occurred as a result of the availability of relatively inexpensive feed grains, proximity to an adequate supply of feeder cattle, and the strong demand for beef. The advantages of confinement include less space per animal, less labor, and economies of scale.

Two types of cattle feeding operations exist, the farmer-feeder who has a feedlot with less than 1000 head capacity and the commercial feedlot which generally has greater than 1000 head and whose business is primarily cattle production. Neither the number of cattle fed or the capacity of the feedlot is the most important criterion to indicate the pollution potential of these two operations. From the pollution standpoint, the important factors are whether runoff retention ponds are used, whether land is available for waste disposal, and how it is used for such disposal. Generally the smaller cattle feeders are farmers and have adequate land available to integrate waste disposal with crop production. The farmer-feeder operations are abundant in the midwestern cornbelt and the large commercial feedlots are numerous in the Texas, Oklahoma, Kansas, Colorado high plains area. The land tributary to the Great Lakes does not feed a significant proportion of the national beef cattle production nor does it have many large feedlots. Individual feedlots of any size can, however, contribute contaminated runoff to surface waters unless adequate controls are utilized.

There are four areas of concentrated cattle feeding in the United States. The area with the most dramatic increase is centered in Texas and Oklahoma where more than 5 million head are handled annually. A second area in the central corn belt feeds about 8 million head and a third area in Nebraska and eastern Colorado feeds about 6 million head annually. The fourth area is in southern California and Arizona which feeds about 3 million head per year. All but the last area have had large increases in fed cattle during the 1960-1970 period.

The large beef feedlots will continue to market a greater proportion of the beef. Feedlots with 1000 head capacity or more account for slightly more than 1% of all the beef cattle feedlots in the United States although they market over 50% of all the beef. The total number of feedlots is decreasing as smaller, inefficient lots go out of business.

The raising of dairy steers in feedlots for meat production is an emerging livestock operation. A dairy beef enterprise offers a supplemental income for dairymen who have labor, feed, and facilities available. With the demand for beef, dairy-beef feedlots should increase in number.

The poultry industry is another example of intensive livestock production. In the major poultry producing regions, most laying hens and broilers are raised in confinement. Large poultry operations are highly mechanized and are able to handle over 100,000 birds per operation. Present poultry management permits the concentration of layers in buildings housing several hundred thousand birds on a site consisting of small acreage.

Over 70 billion eggs were produced in 1970 by about 320 million birds. Egg production is distributed throughout the United States with the greatest increase in production during 1960-70 occurring in the southern and eastern seaboard states. The top five egg producing states in 1970 were California, Georgia, North Carolina, Arkansas, and Pennsylvania. The per capita consumption of shell and processed eggs decreased from 370 in 1955 to about 317 in 1970. Total egg production increased during the same period. Over 90% of U.S. egg production is disposed of as shell eggs. Between 8-10% of the eggs are broken for commercial use as frozen eggs.

The land tributary to the Great Lakes contains significant numbers of egg production operations, especially in Michigan, even though the Great Lakes region is not one of the major national egg production areas.

The swine industry represents about 10% of the total cash farm receipts in the United States ranking only behind beef and dairy cattle among all animal commodities. Swine production is moving toward confined feeding although less than 10% of the hogs were produced in total confinement in 1970. Separate farrowing and finishing herds are common in swine operations having 1000 to 10,000 animals. A few 10,000 to 50,000 hog capacity operations exist. More large, specialized operations will develop incorporating complete feeding, waste treatment, and disposal in a single operation.

About 80% of the U.S. hogs are produced in the North Central states, 10% in the South Central states, 8% in the South Atlantic states, and the rest throughout the country. Six North Central states produce about 60% of the total U.S. hog production. Swine production contains two basic operations--the feeder-pig and growing-finishing operation. These can be under separate or integrated ownership and

management. The swine industry has developed close to the corn production areas of the United States. Because maximum productivity is the objective, swine are fed large amounts of high energy rations with small amounts of roughage.

The trend to confinement animal production operations produce situations that are more analogous to modern industry than they are to the traditionally held picture of the rural small farm. Inputs, i.e., the feed and frequently the raw material such as the young stock, may be purchased elsewhere and brought to the production operation where, under controlled conditions, optimum production of meat, milk or eggs occurs. Dairy and swine production operations remain the closest to the family farm concept. They also are the closest to the desired ecological cycle of crop-animal-waste production with the wastes being returned to the lands producing crops for the animal.

Estimates of the pollutional characteristics of livestock manures are noted in Table 1. These estimates should not be used to indicate the potential water pollution resulting from these wastes but can be used to estimate the general quantity of waste generated by livestock.

2.5 Environmental Concerns

2.5.1 General

The changing agricultural practices noted above have resulted in observed water quality problems. Uncontrolled discharge of wastes from animal confinement operations has been associated with pollution of lakes, rivers, and streams. Surface drainage from cattle feedlots and other areas where animals are concentrated has created considerable national concern. Runoff from manured fields and effluents from animal waste disposal systems also have the potential to cause water quality problems.

Animal wastes may contribute to water pollution in many ways such as: excessive nutrients that increase eutrophication; microorganisms that may impair the use of surface waters for recreational use; impurities in groundwater; contaminants that complicate water treatment; and depletion of dissolved oxygen, causing fish kills.

TABLE 1
QUANTITIES AND CHARACTERISTICS
OF MANURES PRODUCED BY LIVESTOCK (3)
(pounds/animal/day)*

LIVESTOCK	TOTAL MANURE (wet weight)	BOD	SS	NITROGEN	P_2O_5	SODIUM
BULLS COWS (milk) COWS (beef)	90	1.45	1.95	0.33	0.13	0.03
STEERS (beef) HEIFERS (milk) HEIFERS (beef)	50	1.65	2.05	0.16	0.10	0.01
CALVES	25	0.36	0.52	0.08	0.03	0.01
PIGS (feeder)	10	0.38	0.34	0.06	0.04	0.006
SOWS	14	0.41	0.18	0.062	0.042	0.008
SHEEP (ewes)	12	0.32	0.21	0.05	0.03	0.002
SHEEP (lambs)	8	0.22	0.11	0.03	0.02	0.001
HORSES & PONIES	55	1.40	1.90	0.26	0.09	0.01
HENS (layers) CHICKENS (hatchery) TURKEYS (heavy)	0.31	0.025	0.013	0.004	0.0028	0.00025
HENS (pullets) TURKEYS (broiler)	0.16	0.013	0.011	0.0015	0.0008	0.00018
CHICKENS (broiler)	0.09	0.009	0.008	0.0033	0.0002	0.0001
TURKEYS (hatchery)	0.31	0.03	0.02	0.0046	0.00041	0.0004

*one pound = 0.454 kilograms

Detailed information is not available on the number of animal production operations that may have a water pollution problem. General estimates, however, have been made by the U.S. Department of Agriculture using information from several sources (4). In the leading hog producing states, it was estimated that about 22 percent of the operations have problems controlling surface water runoff. In the eighteen major fed beef producing states, 26 percent of the fed beef operations were estimated to have a surface water pollution potential. About 40 percent of the dairy farms in the forty-eight contiguous states were estimated to have need of new runoff control facilities.

Additional data is needed to better understand the comparative role of livestock production operations and their impact on the environment. Neither the quantity of wastes produced, numbers of animals, or the size of the operation are proportional to the amount of pollution that may be generated.

The variability of waste discharges is a complicating factor in assessing such an impact. Animal wastes or wastewaters are not discharged on a regular basis as are industrial or municipal wastes, feedlot runoff is a function of rainfall frequency and intensity, and land disposal of wastes is related to need for disposal and ability to travel on the land. Intermittent discharges from such operations are difficult to define and control.

Data describing potential and actual pollution that can be caused by animal wastes is accumulating. For the nation as a whole, farm animals produce about ten times as much waste material as does the human population. However, the amount of animal wastes that reach ground and surface waters is not well documented. It is incorrect to use the amount of waste defecated by an animal to indicate the actual pollution that may result. Only a small proportion of livestock wastes find their way into surface and groundwaters. The use of any data describing the total wastes from animals serves only to indicate that the potential pollutional problem of animal wastes can be of considerable magnitude and should be considered in overall population control activities.

The heterogeneous characteristics of animal wastes are another difficulty in defining the environmental impact of agricultural production. Methods of storing, handling, and processing livestock wastes affect their characteristics. Animal wastes are produced in a semi-solid form and can be handled in dry, slurry, or liquid form. The form in which the wastes are stored and handled will affect the type of

pollutional constituents and their ability to move to surface and groundwaters.

Nitrogen can serve as an example of the type and changes in characteristics that may exist. In manure, the nitrogen is chiefly in the form of urea, undigested protein, or microbial cells. The urea readily undergoes hydrolysis to ammonium carbonate and can form gaseous ammonia and carbon dioxide. Drying speeds the ammonia loss. Such ammonia release can occur in open feedlots during warm weather. A portion of the ammonia can be nitrified, especially when wastes are aerated, resulting in soluble nitrites and nitrates which are easily transported by surface runoff or soil percolate. Under anaerobic conditions, the nitrogen endproduct will be ammonia which can be released during agitation or oxidized in the soil. Thus depending on how the wastes are handled and stored, the nitrogen content can be in many forms and concentrations. Similar examples can be presented for the carbon fraction of the wastes.

Agricultural, silvicultural, construction, and mining activities can contribute several types of pollutants to surface and groundwaters. A study identifying the nature and extent of these non-point sources of pollutants identified sediment as the major pollutant from these activities (5). The study indicated that the rates of emission of nitrogen and phosphorus are greatest from lands managed for intensive production of crops and livestock. No data on rates of emission other than total sediment were provided in the agricultural component of the study report.

2.5.2 Feedlot Runoff

In most of the United States, animals remain an integral part of cropping systems and their wastes are returned to fields that produce animal feed. The characteristics of the livestock wastes are such that they will usually stay where defecated until an area is cleaned or until runoff washes them away. The water pollution problem associated with livestock wastes primarily is a drainage problem. Runoff from uncovered confinement areas and from land used for disposal of waste occurs during and following rainfall. Pollution caused by feedlot runoff is reduced when the animals are housed as is the case in most poultry and large hog and dairy operations.

Feedlot wastes reach a stream as part of storm runoff. The quantity and pollutional quality of the runoff will

depend upon previous weather conditions, the number of livestock per feedlot area, the method of feedlot operation, soil characteristics, topography of the area, and intensity of rainfall. Information on the characteristics of feedlot runoff is not plentiful due to the variable nature of rainfall-runoff relationships and the logistics of being at the proper sampling location when runoff occurs. In the past few years, studies that have included natural and simulated rainfall events have provided data on the quantity and quality of feedlot runoff under a variety of environmental conditions.

The important environmental factors affecting the quantity and quality of feedlot runoff include rainfall intensity, weather conditions, and feedlot surface. In one of the first detailed studies characterizing feedlot runoff, the greatest pollutant concentrations occurred during warm weather, during periods of low rainfall intensity, and when the manure had been made soluble by water soaking. Ammonia nitrogen concentrations ranged from 16 to 140 mg/l, suspended solids concentrations from 1500 to 12,000 mg/l and COD concentrations from 3000 to 11,000 mg/l in some of the early runoff studies (6). The range of these values is similar to the range obtained in other subsequent studies. Runoff with these constituents can have a detrimental effect on surface waters. Runoff collection ponds can reduce the sediment load in such runoff and have produced an effluent with fairly predictable pollutant concentrations. Liquid from such runoff control ponds commonly is disposed of on adjacent crop land.

The water pollution potential of livestock feedlots is related to the waste production per animal, the number of animals in the confinement unit, days confined, frequency of cleaning, climate, waste characteristics, and waste degradation in the lots. The contribution of feedlot runoff to surface water pollution will be a function of the temperature, magnitude of rainfall, slope of the confinement area, surface area of the feedlot, type of lot surface, and management practices.

2.5.3 Ground Water

Agricultural practices have the capacity to produce pollutants that can enter the groundwater and alter its quality. The magnitude of agricultural contamination of groundwater is difficult to ascertain because of a limited understanding of the amount of pollutants that enter subsur-

face waters under given animal production and waste disposal practices, the changes in groundwater quality caused by such pollutants, and the alterations in pollutant characteristics that occur in the soil. In addition, there is a scarcity of adequate groundwater monitoring programs for these purposes.

Runoff from land and leaching through soil are the two principal means whereby agricultural contaminants can be added to ground and surface water supplies. In contrast to the mobility of nitrate nitrogen in soils, most phosphorus reacts with the soil and little passes through the soil profile into the groundwater. Whereas phosphorus removal from wastewater point sources is receiving emphasis to retard eutrophication, nitrogen may be a more important concern when land disposal of wastes is considered. Phosphorus movement occurs mainly with soil particle movement. Because wind or water erosion is a selective process, preferentially moving smaller, lower specific gravity particles, eroding soil matter can be 2-3 times as rich in nitrogen and phosphorus as the soil from which they were eroded.

Of the rural water supply samples examined in Missouri, 50-75% contained sufficient nitrogen to be of concern in livestock production (7). The main contaminating source both in distribution and concentration was indicated to be waste matter at sites of animal habitation. Soil containing 224 to 4480 kilograms of nitrate-nitrogen per hectare (200 to 4000 pounds/acre) was found below certain feedlots. Standard surface soils contained about 56 to 168 kilograms of nitrate-nitrogen per hectare (50 to 150 pounds/acre). The increased nitrates remained after an area was abandoned from animal use.

Comparative studies in Colorado (8) and Wisconsin (9) have indicated that land use affected the nitrogen content of soils. The highest soil nitrate nitrogen occurred under feedlots. The nitrate nitrogen concentrations in the groundwaters under the lands studied in Colorado ranged from zero to more than 10 mg/l.

A study of nitrate nitrogen in soil under feedlots in Kansas noted accumulations from almost zero to 5100 kg/ha (0 to 4540 lb/acre) in a 4 meter soil (13 feet) profile (10). The amount of accumulation was related directly to the age of the feedlot with the older feedlots having more nitrate nitrogen in the soil profile than did the younger lots. In areas with higher rainfall, nitrate nitrogen accumulated to greater depths than in areas with low rainfall. On soils of fairly constant composition, soil texture

did not effect nitrate-nitrogen distribution patterns beneath feedlots markedly. In soil with varied profiles it was found that as the clay content increased, moisture and nitrate nitrogen content increased. Soil phosphorus analysis indicated little if any movement of phosphorus except on soils with an extremely low exchange capacity. Concentrations of up to 78 mg/l nitrate nitrogen were found in the groundwater below feedlots.

Other studies have indicated the quality of the groundwater in and around animal feeding operations. The groundwater under certain cattle feedlots in the South Platte River valley of Colorado, was observed to contain ammonium nitrogen up to 38 mg/l, organic carbon up to 300 mg/l, and to have had an offensive odor (11). Although much larger amounts of nitrate per unit area were present under feedlots, indications were that irrigated lands contributed more total nitrate to the groundwater since the feedlots occupied a small fraction of this agricultural land area. In Illinois, waste from hog and mink feeding operations contaminated shallow wells causing their abandonment for human and animal use (12). In Kansas, water from a cattle feedlot polluted stream contaminated the well water of a dairy operation. Pollution due to high nitrates caused the death of some cattle (13).

Water from shallow wells is more likely to contain higher nitrate concentrations than water from deeper wells. In particular, shallow wells near barnyards, feeding lots, and manure piles may contain high concentrations. Occasionally, deep drilled wells contain considerable nitrate, either occurring naturally, or entering the well by surface leakage through poor well seals, or from nitrogen-rich deposits within lower soil zones.

Animal production can be the cause of localized surface and groundwater pollution problems. At present, groundwater pollution from animal production sources is, for the most part, below levels that have been demonstrated to cause disease, excessive costs, or aesthetic nuisance.

2.5.4 Land Disposal

The land has been the ultimate disposal point for the wastes of agricultural operations. Agricultural wastes, especially animal wastes, have value in maintaining and improving the soil because of the plant nutrients and organic substances they contain. To avoid pollution caused

by runoff, the wastes should be incorporated with the soil soon after spreading.

Subsoil injection of manure, using mechanical equipment, has shown promise to minimize runoff problems and to eliminate the odor problems that can occur when the wastes are distributed on the surface of the land. The maximum waste application that can be disposed of on the land will depend upon the type of soil, possible buildup of toxic materials in the soil, and potential surface runoff and ground water pollution.

Manure applied to frozen ground can produce contaminants in resultant runoff. Up to 20% of the nitrogen, 12% of the phosphorus and 14% of the potassium in manure applied on frozen ground in the winter was reported lost under conditions favoring maximum early spring runoff (14).

Animal wastes disposed of or defecated on land can contribute constituents to surface streams under natural runoff conditions. The nitrogen and phosphorus content of Nebraska waters was determined and compared to a variety of factors (15). There was little correlation between the N and P concentrations of the waters and the agricultural use of fertilizers. Nutrient levels in the water were better correlated to human and livestock densities and intensity of irrigation than to agronomic production factors.

2.6 Great Lakes Basin

Concerns over pollution and eutrophication problems in the Great Lakes has led the United States and Canada to mutually identify the sources of these problems. Until recently, agricultural inputs into the Great Lakes had not been of serious concern. A report by the International Joint Commission in 1970 on pollution of Lake Erie, Lake Ontario, and the International section of the St. Lawrence River (16) identified wastes discharged by municipalities and industries as the principal causes of the pollution of these waters. Only two paragraphs in the report addressed the question of pollution from land drainage. Those responsible for livestock production were advised to take steps to control animal waste disposal and soil erosion. Agricultural agencies were asked to develop measures to improve practices of soil fertilization and soil conservation to reduce the amount of phosphorus entering the Lakes from the drainage basin.

Increased attention has been given to the impact of agriculture on the pollution problems in the Great Lakes. In 1971, a state-of-the-art document concerning abatement of pollution in the Great Lakes Basin as specifically related to agricultural sources was developed as a joint effort of agencies in the two countries (3). Runoff of pollutants from animal production operations, from associated waste management structures, and from lands used for ultimate livestock waste disposal were considered in the document.

Over 90 million farm animals and birds were estimated to produce in excess of 35 million metric tons (38 million tons) of manure annually in the Canadian portion of the Great Lakes Basin. About 40 percent of these wastes is produced in the Lake Erie watershed. Estimates of the nitrogen and phosphorus produced by the livestock in the Great Lakes Basin are noted in Table 2. The numbers in the Table were determined by different approaches but provide an order-of-magnitude estimate.

No estimate of the magnitude of the pollution hazard resulting from livestock operations in the Great Lakes Basin was obtained in the document. Only total waste production data were presented. Such data do not indicate the actual water pollution that can result since the amount of waste material that does reach surface waters from livestock operations depends upon many factors. A conclusion of the Canadian part of the document was that "it is very difficult to determine with any degree of reliance the amount of pollutant materials from manures that reach the Great Lakes." The practices that have the greatest potential impact on water quality were noted as: winter spreading of manure, storing or accumulating uncovered manure on the ground, spreading manure at very high application rates, and permitting runoff from feeding areas.

The document recommended a survey of actual contributions of all agricultural processes of pollution in the Great Lakes to establish a basis for prescribing means to abate, prevent, or control such sources.

TABLE 2
NITROGEN, PHOSPHORUS, AND BOD IN MANURES PRODUCED BY LIVESTOCK IN THE GREAT LAKES BASIN (ton/year)+(3)

	UNITED STATES PORTION*	CANADIAN PORTION**
LAKE ERIE WATERSHED		
P	15,000	14,200
N	67,000	43,300
BOD	----	458,400
LAKE ONTARIO WATERSHED		
P	9,000	----
N	45,000	----
GREAT LAKES BASIN		
P	----	32,300
N	----	102,100
BOD	----	1,036,700

* produced by cattle, chicken, and hog population
**nutrients available as replacement for commercial fertilizer, assumes some nitrogen lost during handling and storage, and does not include nutrients in animal wastes deposited directly on pasture; BOD represents the BOD in wastes as defecated.
+ one ton = 0.91 metric tons

Section 3

LAND USE

3.1 General

The land use patterns for the area tributary to the Great Lakes have been described in detail (17). The detailed information will not be repeated because it is available and because all of it is not pertinent to this report. However, general information, based on the above and other reports describing the land use in the Great Lakes Basin will be presented as it relates to the production of animals on these lands.

Although there are many major cities within the Great Lakes Basin, the land use is predominantly cropland and woodland (Table 3). Rural land is 84% and urban land about 8.5% of the total area (3). Intensive animal production will occur on only a small fraction of the rural land area. The major descriptions of land use in Table 3 are inadequate to identify the land used for livestock production.

Animal production on typical farms in the Basin generally occurs as a part of a combined crop-livestock production system. Wastes are distributed on adjacent cropland for disposal. It may be possible to use the values for pasture land as an upper limit estimate of the relative water pollution potential of animal production in the Basin as compared to other agricultural land use. Using such a percentage as an estimate, runoff from pastures is only about 5-14% of that from other agricultural lands in the Basin. The water pollution potential from animal production areas in the Basin may differ from these percentages because runoff from these areas generally has a higher concentration of contaminants than does runoff from crop and woodland and because the land described as pasture may not be an adequate estimate of the land used for livestock production.

3.2 Livestock Producing Farms

All types of livestock are produced in the Basin. Dairying is the major livestock operation in the Basin, with the distribution of livestock operations in the Basin being: dairy farms-30%, other livestock farms-8.6%, and poultry farms-2.1% of the farms in the Basin (17).

TABLE 3
DISTRIBUTION OF MAJOR LAND USE
IN THE GREAT LAKES BASIN

LAND USE	PERCENT OF TOTAL LAND IN FARMS	GENERAL LAND USE	PERCENT OF RURAL LAND
CROPLAND - HARVESTED	53%	CROPLAND	40%
WOODLAND NOT PASTURED	10.6%	PASTURE	5%
CROPLAND - IDLE	8.9%	FOREST	48%
CROPLAND - PASTURED	7.5%	OTHER	7%
OTHER LAND	7.5%		
OTHER PASTURE	6.4%	DATE	1967
WOODLAND PASTURE	6.1%		
DATE	1964	REFERENCE	3
REFERENCE	17		

The total number of farms producing livestock in the counties with land tributary to the Great Lakes are shown in Table 4. These farms are those at least ten acres in size and with sales of at least $2500 in the year previous to the census. There were a number of smaller farms in each of the counties but it was felt that the farms noted in Table 4 were the significant animal producers in the Basin.

The number of animals per farm in each of the states adjacent to the Great Lakes were not very consistent (Table 5) among the respective animal categories. Such data represents the average in each state. However, in the absence of more specific data, it is reasonable to assume that the values in Table 5 are roughly representative of farms in the Basin.

The Great Lakes states are not the major beef producing states in the United States. There are a number of cattle feedlots in these states, with some lots having a capacity greater than 1000 head (Table 6). The number of beef cattle feedlots in the Great Lakes Basin is unknown due to lack of more specific data. Based on the distribution of land in the Basin as a percentage of the total land area in each state, it is assumed that the major beef cattle feedlots in the Basin exist in Michigan.

The majority of the beef cattle feeding operations in these states are less than 1000 head and undoubtedly can be classified as a farmer-feeder operation. Such operations generally are a combined livestock-feed production operation in which the manure is disposed of on cropland. They also are not likely to be enclosed and runoff with feedlot characteristics is possible. On the other hand, the small feeding operations are not as likely to have the animals as concentrated which in turn can reduce the pollutional characteristics of the runoff.

3.3 Livestock Distribution

Although various types of livestock are produced throughout the Great Lakes Basin, the major livestock production occurs where satisfactory cropland is available for forage and where the production facilities have access to major markets (Table 7). In addition to dairy and beef cattle, hogs and pigs, poultry and sheep and lambs, some turkey as well as duck and geese production occurs in the Basin. The counties in which large scale turkey and duck production

TABLE 4
TOTAL NUMBER OF FARMS PRODUCING LIVESTOCK IN THE COUNTIES WITH LAND TRIBUTARY TO THE GREAT LAKES*

STATE	NUMBER OF FARMS		
	CATTLE AND CALVES	HOGS AND PIGS	POULTRY
MINNESOTA	1,790	510	322
WISCONSIN	21,150	6,345	4,810
ILLINOIS	2,280	1,020	770
INDIANA	6,165	4,290	1,640
MICHIGAN	24,910	7,860	5,270
OHIO	10,655	7,175	3,255
PENNSYLVANIA	685	73	100
NEW YORK	14,990	1,485	2,020

*compiled from data in the 1969 Census of Agriculture (18)

TABLE 5
AVERAGE ANIMALS PER FARM IN GREAT LAKE BASIN STATES*

STATE	ANIMALS PER FARM			
	CATTLE AND CALVES	HOGS AND PIGS	POULTRY	SHEEP AND LAMBS
MINNESOTA	56	93	650	58
WISCONSIN	56	72	340	39
ILLINOIS	56	157	530	32
INDIANA	44	144	1,730	32
MICHIGAN	53	84	1,350	90
OHIO	47	102	1,120	56
PENNSYLVANIA	49	48	1,790	44
NEW YORK	62	28	3,220	67

*computed from data in the 1969 Census of Agriculture (18)

TABLE 6
NUMBER AND CAPACITY OF CATTLE FEEDLOTS IN THE GREAT LAKE BASIN STATES - 1971 (19)

	NUMBER OF LOTS				
	HEAD OF FEEDLOT CAPACITY				
STATE	LESS THAN 1,000	1,000-1,999	2,000-3,999	4,000-7,999	8,000+
MINNESOTA	15,960	35	5	-	-
WISCONSIN	7,293	7*	-	-	-
ILLINOIS	17,943	44	8	5*	-
INDIANA	12,476	19	5*	-	-
MICHIGAN	1,671	20	9	-	-
OHIO	8,972	25	3*	-	-
PENNSYLVANIA	5,997	3*	-	-	-
NEW YORK	NOT AMONG THE 23 MAJOR FEEDING STATES				

*lots from larger size groups are included to avoid disclosing individual operations

TABLE 7
MAJOR DISTRIBUTION OF LIVESTOCK IN THE
GREAT LAKES BASIN (17)

LIVESTOCK	GEOGRAPHICAL AREA IN BASIN	GREAT LAKES SUB-AREA	PERCENT OF BASIN PRODUCTION	DRAINAGE TO LAKE
DAIRY CATTLE	UPPER WISCONSIN	2.1	25%	MICHIGAN
	LOWER WISCONSIN	2.2	12%	MICHIGAN
	LOWER MICHIGAN	2.3	11%	MICHIGAN
	WESTERN NEW YORK	5.1	10%	ONTARIO
HOGS AND PIGS	WESTERN OHIO	4.2	31%	ERIE
	LOWER WISCONSIN	2.2	24%	MICHIGAN
	LOWER MICHIGAN	2.3	17%	MICHIGAN
	UPPER MICHIGAN	2.4	15%	MICHIGAN
POULTRY	LOWER WISCONSIN	2.2	--	MICHIGAN
	WESTERN OHIO	4.2	--	ERIE
SHEEP AND LAMBS	WESTERN OHIO	4.2	31%	ERIE
	LOWER WISCONSIN	2.2	25%	MICHIGAN

occur are (18):

- turkeys - Wisconsin: Douglas; Michigan: Allegan, Barry, Cass, Ottawa

- ducks - Wisconsin: Racine; Ohio: Sandusky, Hancock

Based upon the major categories of livestock, the major drainage from animal production operations in the Basin appears to enter Lake Michigan from Wisconsin and lower Michigan and to enter the western end of Lake Erie from Ohio (Table 7). About 50% of the dairy cattle and about 56% of the hogs and pigs are in the Lake Michigan drainage basin.

Comparatively few livestock operations are in the Lake Superior drainage Basin and therefore potential pollution from animal operations into Lake Superior appears minimal. Because of the small amount of land in Illinois and Indiana that is tributary to Lake Michigan, animal production systems in these states are not major contributions to potential pollution from animal operations in the Great Lakes Basin.

The numbers of cattle and calves and hogs and pigs in the Great Lakes Basin states increased slightly in the 1970-73 period. The numbers of sheep and lambs in the same period decreased in all these states (Table 8). The change was less than ten percent in any state.

The extent of potential water pollution problems from livestock operations depends upon a number of factors which include: a) number of animals per operation, b) acreage per farm, and c) general production technology. Changing farm technology encourages more intensive agriculture, concentration of livestock production, intensive use of land resources, and large-scale livestock production. The large-scale operations frequently are given the largest publicity both from the production and potential pollution standpoint. However, the major proportion of United States livestock production results from many hundred thousand small farms.

The less intensive family farm will continue to produce most of the livestock in the Great Lakes Basin for the forseeable future. The family farm has been defined as that which an operator and an average size family can manage and work mainly on their own (21). Throughout the United States, about 95 percent of the farms are family farms. In the Lake states, 98% of the farms are family farms and produce about

TABLE 8
INVENTORY OF MEAT ANIMALS IN THE
GREAT LAKES BASIN STATES (20)(1,000 HEAD)

STATE	CATTLE AND CALVES** NUMBER	CHANGE*	HOGS AND PIGS*** NUMBER	CHANGE*	SHEEP AND LAMBS** NUMBER	CHANGE*
MINNESOTA	4,038	+	3,650	+	437	-
WISCONSIN	4,283	+	1,475	-	124	-
ILLINOIS	3,330	0	7,000	+	275	-
INDIANA	2,025	+	4,950	+	235	-
MICHIGAN	1,576	+	751	+	221	-
OHIO	2,134	+	2,402	-	627	-
PENNSYLVANIA	1,745	-	583	+	144	-
NEW YORK	1,764	-	89	-	98	-

* indicates change in inventory over similar inventory period three years earlier; + = increase, - = decrease, 0 = no change
** inventory date - January 1, 1973
***inventory date - December 1, 1972

90 percent of all farm products sold. In the content of this report, family farms are considered general animal production operations and not intensive animal feedlots.

The average number of animals per farm in the Great Lake states has been noted in Table 5. In the counties tributary to the Great Lakes, about 63% of the farms have less than 50 head of cattle and calves, about 60% of the farms have less than 50 head of hogs and pigs, and about 90% of the farms have less than 3200 poultry (17). Most livestock enterprises in the Great Lake states are of the multi-crop-livestock type.

The numbers of livestock in a few of the Great Lake states provide more detailed insight on typical sizes of operations. In Illinois, a major hog production state, half of the producers raised fewer than 150 head annually. Cattle feeding is important in Illinois, however over 98% of the producers sold fewer than 500 head in 1969. Over 90% of the dairy farms in Illinois had less than 50 head (21).

The density of livestock per acre also provides information useful in estimating the water pollution from livestock operations. Generally, the family farm has more land area per animal and since these are integrated crop-livestock operations, the animal wastes are returned to the crop land. In Illinois, the density of hogs produced annually ranged from 7.4 to 14.8 head per hectare (3 to 6 head per acre) of total farmland with greater densities on farms on which more grain is fed than produced. The latter type of farm is more typical of intensive animal operations. In the Great Lakes states, dairying usually is associated with farmland because relatively little forage is harvested for sale. In Michigan in 1970, the cropland per dairy cow ranged from 1.85 hectares (4.6 acres) per cow on smaller dairy farms to 1.2 to 1.4 hectares/cow (3.0 to 3.5 acres/cow) on farms with larger herds, generally over 90 cows/farm (21). In Wisconsin counties tributary to Lake Michigan the dairy cow density ranged from 0.8 to 2.6 hectares (2.0 to 6.4 acres) /cow, the hog and pig density from 0.4 to 1.6 hectares (1.0 to 4.0 acres)/head, and the poultry operations generally less than 22,000 birds per hectare (9,000 birds/acre) for even the large commercial operations (9).

Although most livestock production in the Great Lakes Basin remains land based, some farmers have moved livestock off pasture into more concentrated production operations. This shift permits more mechanization and an increased economic efficiency. Most beef cattle are fattened in a feedlot rather than on pasture. Many large hog enterprises use

some form of confinement although most production still takes place on pasture or in combination drylot-pasture facilities. Many dairy operations have moved to drylots using harvested feeds.

The intensive animal feedlot operations may not be closely integrated with crop production, the animals may be in enclosed or open confinement so that the wastes cannot be absorbed by the immediate environment, and runoff similar to that from the beef cattle feedlots in the western high plains area can occur. The water pollution potential due to runoff is greater from intensive animal feedlots than from family farm livestock crop operations. Because of this potential, it is useful to identify where the intensive animal feedlots are in the Great Lakes Basin.

Operationally, intensive animal feedlots can be defined as a facility capable of holding animals on land not used for the growing of crops or vegetation. The concentration of animals is high enough that crops cannot grow.

In terms of animal numbers, the following limits were used to define an intensive animal feedlot: dairy farm - 100 or more head; hogs or pigs - 200 or more capacity; poultry - 10,000 or more birds, and beef cattle - 1000 head or more. These numerical limits were based upon what was felt to be the size of an operation that would operate at a respectable profit and that would be a large, single enterprise operation.

Figures 1-4 identify the intensive animal feedlots in the Great Lakes Basin. No such feedlots exist in Plan Area No. 1 which includes land draining to Lake Superior. Data was based on information in the 1969 Census of Agriculture (18).

Maps showing the distribution of intensive cattle, hog, and poultry facilities in counties tributary to the Great Lakes are presented in the Appendix, Figures A-1 through A-9. The intensive animal feedlots are in the same areas where most of the livestock is produced. Based upon data in Table 6, the intensive beef cattle feedlots in the Great Lakes Basin are in Michigan and possibly Ohio. The locations of the large cattle operations (Appendix) may be either dairy or beef operations since the data is not separated as to type of cattle. They are more likely to be dairy operations because of the small numbers of large beef cattle operations in the Basin.

In summary, large, intensive, livestock production operations are not common in the Great Lakes Basin. The majority

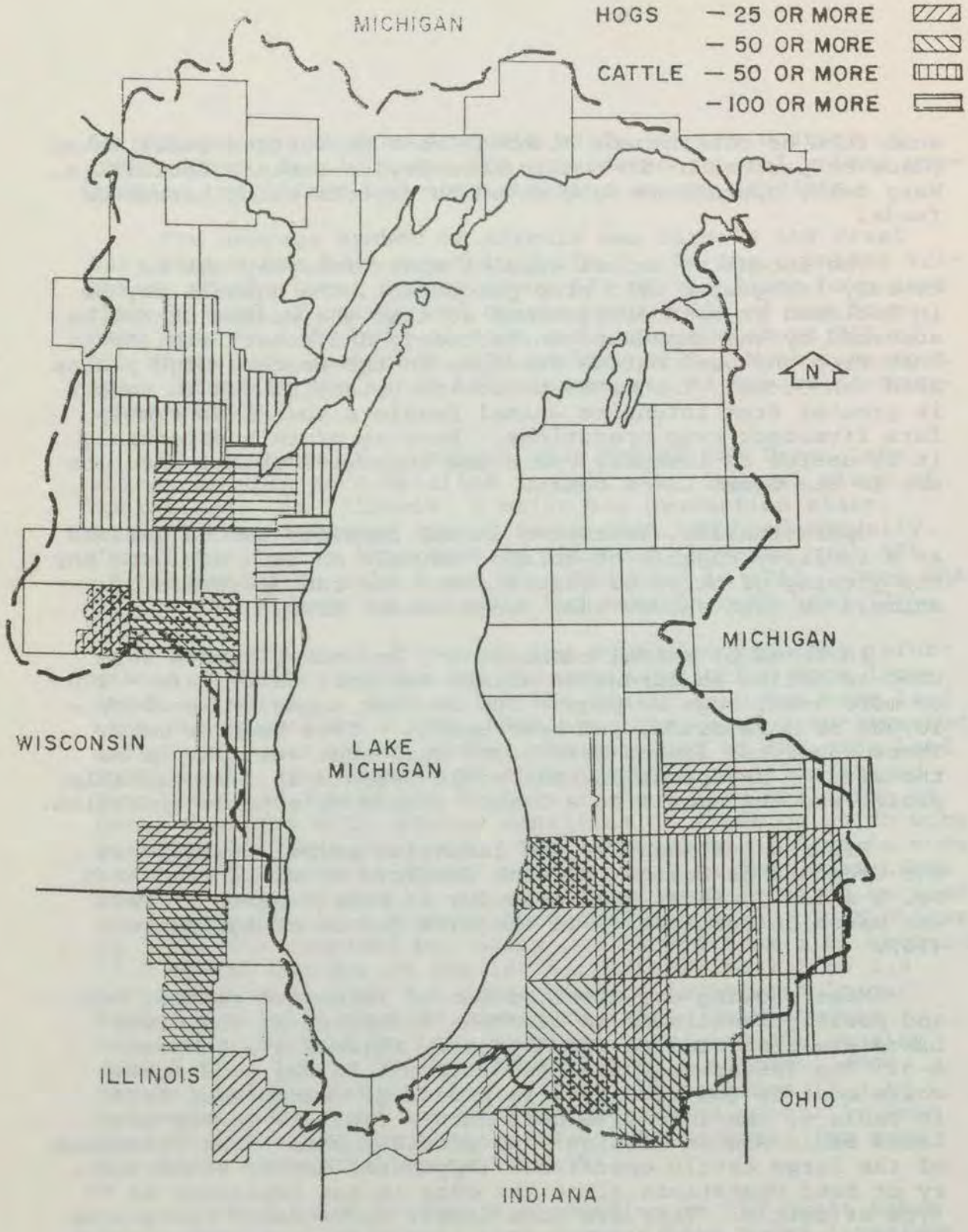


FIGURE 1
DISTRIBUTION OF INTENSIVE
LIVESTOCK OPERATIONS IN COUNTIES
TRIBUTARY TO LAKE MICHIGAN

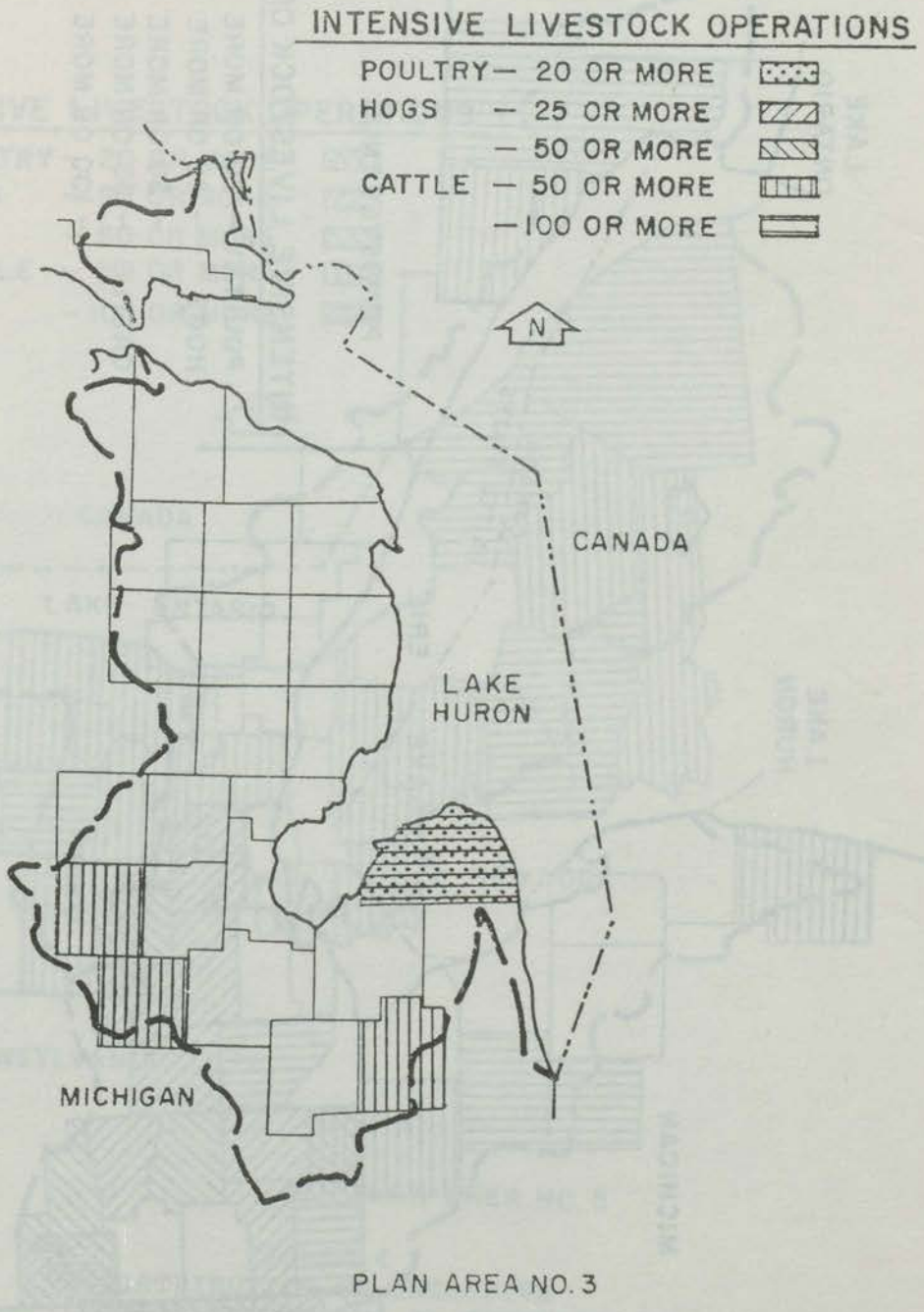


FIGURE 2
DISTRIBUTION OF INTENSIVE
LIVESTOCK OPERATIONS IN COUNTIES
TRIBUTARY TO LAKE HURON

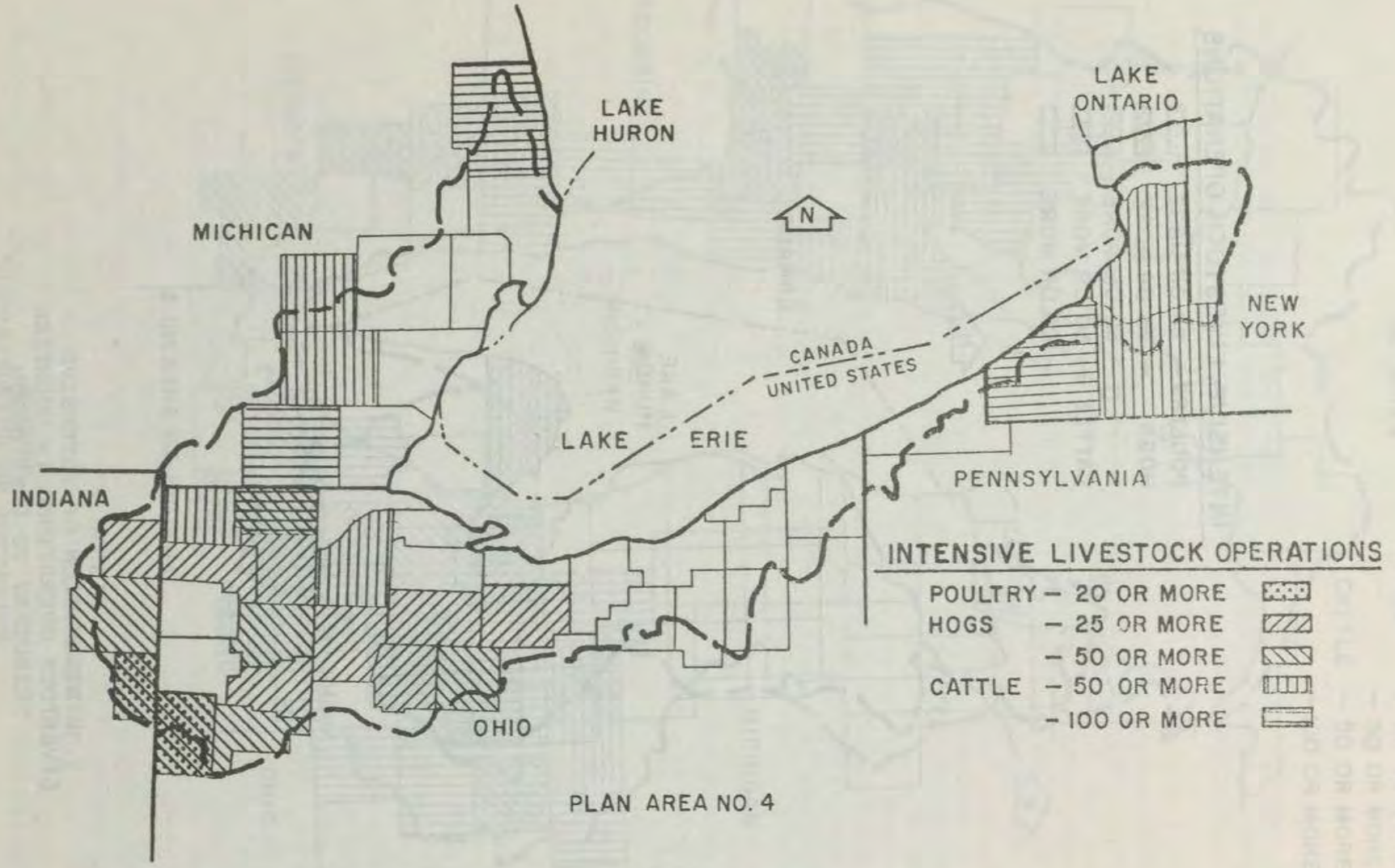
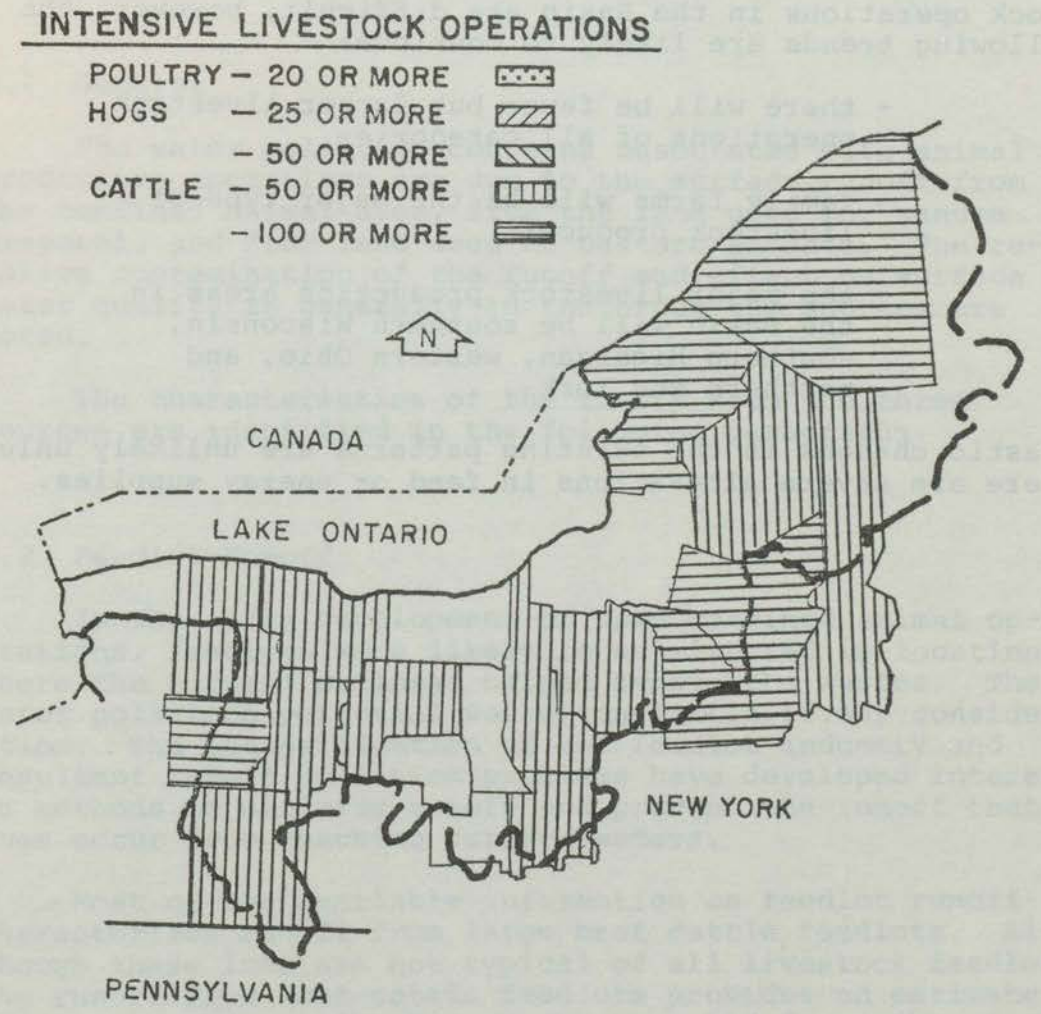


FIGURE 3.
DISTRIBUTION OF INTENSIVE
LIVESTOCK OPERATIONS IN COUNTIES
TRIBUTARY TO LAKE ERIE



PLAN AREA NO. 5

FIGURE 4
DISTRIBUTION OF INTENSIVE
LIVESTOCK OPERATIONS IN COUNTIES
TRIBUTARY TO LAKE ONTARIO

of livestock operations are family farms and the average animal distribution per land area is much greater than that found in the commercial beef cattle feedlots in the mid- and far west. Accurate future predictions of size of livestock operations in the Basin are difficult, however, the following trends are likely to continue:

- there will be fewer but larger livestock operations of all categories
- family farms will be the major type of livestock producer
- the major livestock production areas in the Basin will be southern Wisconsin, southern Michigan, western Ohio, and northern New York

Drastic changes in the existing patterns are unlikely unless there are severe alterations in feed or energy supplies.

Section 4

SURFACE AND GROUNDWATER CONTAMINATION

4.1 General

The water pollution concerns associated with animal production operations are due to the surface runoff from the confined animal area, from the land used for manure disposal, and from land used to pasture animals. The relative contamination of the runoff and effect on surface water quality is generally in the order the sources are noted.

The characteristics of the runoff from the three sources are identified in the following paragraphs.

4.2 Feedlot Runoff

In the early development of open confined animal operations, feedlots were likely to be situated in locations where the natural drainage helped remove the wastes. The water pollution potential was given little if any consideration. The intensification of the feedlot industry and resultant runoff pollution problems have developed interest in methods to minimize runoff and prevent the runoff that does occur from reaching surface waters.

Most of the available information on feedlot runoff characterizes runoff from large beef cattle feedlots. Although these lots are not typical of all livestock feedlots, the runoff from beef cattle feedlots provides an estimate of the type of runoff that may be expected from other confined animal operations. The quantity and quality of feedlot runoff will depend upon prior soil water content, the number of animals per feedlot area, the method of feedlot operation, soil characteristics, topography of the area, and intensity of rainfall.

The degree of manure decomposition that takes place in a feedlot is dependent upon temperature and moisture conditions and in turn affects the characteristics of the runoff. The greater the moisture content of the wastes, the greater the degree of waste solubilization and the greater the amount of soluble constituents in subsequent runoff. Because of a decrease in bacterial action and waste decompo-

sition in the winter, there will be a greater accumulation of wastes in an open feedlot during this period than during the rest of the year. While the manure is decomposing on the lot, it is constantly being mixed with fresh wastes and with soil if the feedlot is unsurfaced. The characteristics of the actual waste in a feedlot will be a function of these physical and biochemical changes.

The highest concentrations of pollutants in feedlot runoff occur in the initial runoff and decrease to a lower, more uniform level as runoff continues. This lower level results from the gradual dissolving of materials in the surface layer of manure on the lots. Feedlot runoff quality is not sensitive to the quantity of manure on the lot. Once the feedlot surface is covered, the depth of manure is not an important parameter of water quality. The important variables that do affect the pollutional constituents in feedlot runoff include rainfall intensity, prior water content of the manure pack, and the type of feedlot surface.

Data from a number of studies have described the magnitude and variability of constituents of feedlot runoff. A study in Texas found feedlot runoff to contain from 500 to 3300 mg/l of BOD, and 3400 to 13,400 mg/l of suspended solids (22). Another report from Texas noted that cattle feedlot runoff had characteristics of 1000-16,800 mg/l BOD, 2900-48,000 mg/l COD, 6-800 mg/l organic nitrogen, 2-770 mg/l ammonia nitrogen, 0-1270 mg/l nitrate nitrogen, and 70-2400 mg/l alkalinity (23). Feedlot runoff data from Colorado indicated BOD concentrations from 300 to 6000 mg/l, volatile solids from 1000 to 7000 mg/l, and settleable suspended solids from 0.1 to 30 ml/l (24). Data on runoff from feedlots in Kansas have comparable values: BOD - 100-11,000 mg/l, COD - 4,000-40,000 mg/l, total solids - 10,000-25,000 mg/l and Kjeldahl nitrogen from 200-450 mg/l (25, 26). These concentrations represent the quality of runoff as it leaves the feedlot. The significance of this problem is the highly variable nature of the runoff and its slug effect on a stream. In a stream, the concentrations in the feedlot runoff will be diluted by the flow from other portions of the contributing watershed.

Although climatic conditions and size of beef feedlots are different in Canada, studies on the quality of feedlot runoff in Ontario have produced data similar to that noted in the United States. Runoff from the Canadian feedlots contained from 800 to 7500 mg/l BOD_5, 265 to 3400 mg/l total Kjeldahl nitrogen, and 93 to 2180 mg/l phosphorus as P_2O_5. The events causing the runoff were 0.76 to 2.54 cm (0.3 to 1 inch) rains with the higher concentrations occurring shortly after the larger rains (27).

In the semi-arid regions of the southwest, cattle wastes are dehydrated in a short time and remain that way until wetted by precipitation. The wetting may reconstitute the wastes to almost original composition. In more humid climates, the wastes may remain moist for longer periods of time before natural drying occurs. The longer the wastes remain moist, the greater the opportunity for bacterial action and solubilization of the solid matter. From spring through fall, the greatest pollutant concentrations are obtained during warm weather, during periods of low rainfall intensity, and when the manure had been made soluble by water soaking. Runoff from winter thawing conditions has contained higher waste concentrations than that contained in the runoff caused by rainfall under warmer conditions (28). Runoff from high density lots (9.3 m^2/head) (100 ft^2/head) contained 130-170% greater runoff quantities and 4-5 times more total solids than did runoff from lower (18.6 m^2/head) (200 ft^2/head) density lots.

Pollution from an uncovered livestock area, such as feedlot, is related to the fraction of precipitation that becomes runoff and reaches surface streams. Only after a portion of the rainfall soaks into the manure does runoff occur. This fraction will depend upon previous weather conditions.

Available information on rainfall-runoff relationships observed at small surfaced and unsurfaced feedlots indicated (29) that on surfaced feedlots,

$$\text{Runoff} = -0.34 + 0.945 \text{ Rainfall}$$

Similar coefficients were observed with relationships from unsurfaced feedlots.

Other data indicated the following average relationship for runoff at cattle feedlots (28) to be:

$$\text{Runoff} = -0.135 + 0.53 \text{ Rainfall}$$

Data from other studies indicated that from 0.56 to 1.3 cm (0.22 to 0.5 inches) of precipitation was needed before runoff occurred and that from 0.45 to 0.68 of the precipitation became runoff.

These rainfall-runoff relationships also indicated that after a minimum amount of rainfall, most of the rainfall ended up running off the feedlot. A dry, hard layer occurs several inches below the manure surface that will resist water penetration unless disturbed.

The above data suggest that infiltration to groundwaters can be neglected in determining the total runoff from operating feedlots. This point has been verified in studies in Nebraska (30). When feedlots are abandoned, the accumulated wastes may remain, the hard crust break up, and greater downward motion of pollutants may occur.

Based upon detailed studies, the annual average BOD discharged per hectare of feedlot surface was estimated to be 2800 kg/ha (2500 pounds/acre) for concrete feedlots and 1350 kg/ha (1200 pounds/acre) for nonsurfaced feedlots (6). On this basis the annual average pollutional population equivalent per hectare of feedlot was estimated to be 100 (40 per acre) for flat, surfaced lots and 50 (20 per acre) for similar non-surfaced feedlots.

Data from South Dakota (31) indicated that only a small proportion of the wastes deposited on a feedlot is removed by runoff: TKN - 0.1 to 6.6%, BOD - 0.5 to 3.5%, COD - 0.5 to 2.8%, and phosphate - 0.8 to 12.5%. An estimate of contaminant load per acre of feedlots for those which were studied are noted in Table 9. The data in Table 9 are in general agreement with the information noted above.

Range fattened cattle represent a smaller runoff pollution problem than do feedlot cattle since they are more widely distributed over the land. As the density of animals per surface area decreases, the wastes are less concentrated and nature can absorb more of the wastes. The beef cattle operations in the Great Lakes states are smaller and generally have fewer animals per acre than do the feedlots in the major beef cattle feeding states.

The quantity of waste defecated by the cattle or any animal in a confined feedlot should not be used to indicate the pollution that will reach surface streams since only a small percentage, less than about 5% of the oxygen demanding materials in the waste on a cattle feedlot, are removed by runoff. This percentage may be larger where feedlots are subject to poor management or situated on slopes adjacent to streams.

The pollutional effect of feedlot runoff occurs on an intermittent basis since it is related to precipitation events. As a result, the runoff has a potentially greater adverse effect on surface waters than if it were released at a continuous lower rate.

Even feedlot runoff retained in a runoff collection pond is not of a quality that should be released to surface

TABLE 9
RUNOFF CARRIED WASTE PER ACRE OF FEEDLOT PER YEAR
MEAN VALUES (31) (lb/acre/yr)*

TYPE OF OPERATION	TKN	COD	BOD	TOTAL PHOSPHATE	TOTAL SOLIDS	VOLATILE SOLIDS
COMMERCIAL BEEF FINISHING	982	14,316	3,369	298	18,315	9,065
FAT LAMB FINISHING	41	643	120	25	1,428	718
FAT LAMB FINISHING	73	1,245	168	31	2,207	991
DAIRY CONFINEMENT	1,012	12,770	2,428	391	20,678	7,552
BEEF AND SHEEP FEEDING	628	8,116	1,820	1,276	10,942	6 620
BEEF FEEDING	89	1,543	522	25	2,743	1,375

average precipitation - 20-25"/yr; average feedlot slopes - 2-15%

*1.0 lb/acre = 1.12 kg/ha

waters except in extreme storm events. Discharge of large quantities of runoff collected in retention ponds to a 18 hectare (45 acre) flood control reservoir killed essentially all the game fish in the reservoir (32). The cause of the kill was zero dissolved oxygen and high free ammonia concentration.

While much of the data describing feedlot runoff characteristics provides information on the characteristics of individual samples or runoff events, few studies are comprehensive enough to describe the runoff in terms of per unit values such as amount of pollutants per head of animal or per area of feedlot. Some data of this nature has been presented in Table 9. Other information from studies where adequate data were available to estimate a potential area yield rate, such as kilograms of pollutant per hectare of feedlot per year (kg/ha/yr), are presented in Table 10. Additional information on the constituents of beef cattle feedlot runoff is shown in Table 11 and a comparative summary of beef feedlot runoff characteristics is presented in Table 12.

The scarcity of definitive data and the variability of the available data indicates extreme caution in attempting to extrapolate the area yield rate values in Tables 9 and 10 to indicate the potential pollution from other cattle feedlots or confined animal operations. Although it is logical that the amount of pollutants should be related to the land area occupied by the animals and to the number of animals per land area, the characteristics of feedlot runoff are more a function of precipitation, temperature, and other environmental conditions.

4.3 Barnlot Runoff

Runoff from the large animal production operations has caused considerable concern. Characteristics of such runoff and potential control measures have been the subject of many investigations. However there are larger numbers of small livestock operations from which runoff can contaminate surface waters. Only a few studies have been conducted to identify the runoff characteristics from such operations.

At these small operations, the animals may be housed for feeding, watering, and during cold weather. They may have access to an adjacent confined unenclosed area for exercise during the day. Cattle may be on pasture during

TABLE 10
BEEF CATTLE FEEDLOT AREA YIELD RATES
(kg/ha/yr)*(28)

CONDITION	TOTAL SOLIDS	VOLATILE SOLIDS	TOTAL NITROGEN	TOTAL PHOSPHORUS
-snowmelt runoff				
-100 ft^2/head-1969	60,000	31,000	1,600	620
-1970	1,100	770	100	10
-200 ft^2/head-1969	14,000	7,200	450	200
-1970	640	450	360	60
-rainfall runoff				
-100 ft^2/head-1969	7,500	3,400	330	22
-1970	27,000	16,000	900	130
-200 ft^2/head-1969	7,300	3,400	220	33
-1970	18,000	10,000	350	70

*1.0 kg/ha = 0.89 lb/acre

TABLE 11
MAJOR IONS IN CATTLE FEEDLOT RUNOFF
(mg/l)(28)

ION	MEAN	RANGE
Na	840	40-2750
K	2520	50-8250
Ca	790	75-3460
Mg	490	30-2350
Zn	110	1-415
Cu	7.6	0.6-28
Fe	765	24-4170
Mn	27	0.5-146

TABLE 12
BEEF CATTLE FEEDLOT RUNOFF CHARACTERISTICS (mg/l)

Location	Constituent							
	COD	BOD	TKN	NH_4-N	NO_3-N	Total N	Total P	Ref.
Nebraska								
-snowmelt runoff	41,000 (14,100-77,000)*	---	---	780 (6-2020)	17 (0-280)	2,100 (190-6530)	290 (5-920)	28
-rainfall runoff	3,100 (1,300-8,200)	---	---	140 (2-1240)	10 (0-220)	920 (11-8590)	360 (4-5200)	28
Texas								
-dirt lots	9,500 (2,900-28,000)	1,460 (1,010-2,200)	128 (9-280)	56 (2-85)	23 (0-103)	---	---	23
-concrete lots	21,500 (8,400-32,800)	8,000 (3,300-12,700)	550 (70-1070)	350 (33-775)	220 (0-880)	---	---	23
Colorado	----	300-6000	---	---	---	---	---	24
Kansas	4,000-40,000	1,000-11,000	200-450	---	---	---	---	29

*data in parentheses indicate the reported range of data

the warmer weather. Most of the manure may be removed and spread on cropland. The runoff from the confined open lot can be an item of concern.

Data from a small unpaved barnlot in Ohio (33) collected over a 17 month period indicated that runoff usually occurred when rainfall exceeded 1.3 cm (0.5 in.). The concentrations of BOD and COD within a runoff event were variable. BOD concentrations ranged from 9 to 566 mg/l while COD concentrations ranged from 350 to 5650 mg/l for the events that were sampled. On a rainfall event basis, the BOD yield ranged from 0.06 to 63 kg BOD/ha (0.05 to 56 pounds/acre). The BOD concentration and yield was larger in the winter and smaller in the summer.

4.4 Manure Storage

Where manure cannot be disposed on land in all seasons, such as in the winter, it can be stacked and stored until conditions permit land disposal and integration with crop production. During storage, seepage from the stored manure can occur and can be a source of pollution. Table 13 indicates the characteristics of seepage from stacked dairy manure. Although the volume of seepage is small, the quantity of contaminants is not insignificant.

The stored manure seepage can be controlled by retention ponds and distribution on cropland in a non-pollutional manner. Even though manure seepage can occur in a large number of locations throughout the country, and as such approximates a non-point source, it can be considered as a controllable source of pollution.

4.5 Land Used for Manure Disposal

The land is the most acceptable point of disposal for animal wastes. At most livestock operations, the animal manure is returned to adjacent cropland. At a few operations, the land may be used primarily for manure disposal with crop production not being an important factor. In either case runoff from these lands can contain contaminants from the manure. In addition, the possibility of groundwater contamination needs to be considered.

TABLE 13
CHARACTERISTICS OF SEEPAGE FROM STACKED
DAIRY CATTLE MANURE AND BEDDING (34)

PARAMETER	WINTER AVERAGE	WINTER RANGE	SUMMER AVERAGE	SUMMER RANGE
TOTAL SOLIDS (%)	2.8	1.8-4.3	2.3	1.7-2.9
VOLATILE SOLIDS (%TS)	55	52-59	53	50-58
SUSPENDED SOLIDS (%)	0.35	0.2-0.8	0.24	0.2-0.3
BOD-mg/l	13,800	4,200-31,000	10,300	4,400-21,700
COD-mg/l	31,500	21,000-41,000	25,900	16,400-33,300
TOTAL N-mg/l as N	2,350	1,500-2,900	1,800	1,200-2,770
NH_4-N-mg/l	1,600	980-1,980	1,330	780-2,200
TOTAL P-mg/l as P	280	64-560	190	90-340
POTASSIUM-mg/l as K	4,700	3,000-7,200	3,900	3,000-4,900
TOTAL PRECIPITATION*-inches	15.0		9.4	
SEEPAGE VOLUME**-(gal/cow/day)	3.0		1.2	

* 1 inch = 2.54 cm

** 1 gallon = 3.79 liters

4.5.1 Manure Application Rates

Although guidelines are available for the amount of manure to be disposed of per unit of land, additional criteria need to be developed to provide a better understanding of the limitations of a soil to accept wastes. The maximum waste application rates will depend upon the type of soil, potential groundwater problems, and available crop growth that can remove the added nutrients. Conservatively, the amount of nutrients in wastes disposed of on land should be no more than the amount that can be removed by the crop, weeds, or forest grown on the land. There are, however, mechanisms of nitrogen loss that decrease the potential of manure nitrogen to increase the nitrate concentration of soil water.

The continued application of manure to land at rates which supply nitrogen in excess of crop requirements may result in a loss of nitrate nitrogen from the root zone of the crop. This loss represents a potential groundwater problem. Nitrogen in manure sources may not be as easily lost in soil percolate as that from fertilizer sources. Manure nitrogen conversion to nitrate is dependent upon biological factors which closely parallel those governing plant growth. This can allow greater opportunity for plant uptake before leaching occurs.

Nitrogen lost by denitrification in soils is not a pollution problem as the primary product is nitrogen gas (N_2) which is the main constituent of the atmosphere. However, this nitrogen is no longer available for crop growth, and loss by denitrification is not desirable unless the nitrogen is in excess and has to be disposed of in a non-pollutional manner. Losses of nitrogen by volatilization of ammonia also can occur but cannot be considered as non-pollutional in the same way as denitrification losses.

When animal wastes are applied to land and not plowed into the soil quickly, some of the ammonia nitrogen will volatilize. About 50% of the ammonia nitrogen in poultry manure will volatilize upon air drying. A similar fraction of the ammonia nitrogen will be lost from other animal manures when left to dry on the surface of the land. This nitrogen loss has a positive benefit since it will decrease the potential of increased nitrate nitrogen in the soil water. However, it also has a negative benefit for those farmers who wish to make maximum use of the nutrients in the manure for crop production. The volatilized ammonia will return to the earth surface at another location,

possibly contributing to increased nitrogen in surface waters. Incorporation of the manure into soil soon after it is spread will reduce the ammonia loss due to volatilization but can increase the amount of nitrate nitrogen that will be formed and moved into the soil profile by percolating water.

The application of poultry manure to the soil in light but frequent applications resulted in less nitrate accumulation in the soil profile then did less frequent, heavier applications (35). Thus to minimize increased nitrate concentrations in soil water, manure should be applied as frequent light applications.

It is difficult to identify a particular rate of manure nitrogen application which is excessive. Important factors in determining the appropriate rate include the time of application, the crop being grown, type of manure applied, soil characteristics, and general weather conditions. Data is becoming available that permits estimates to be made of appropriate application rates.

Zero to 3900 kilograms of manure nitrogen in dairy manure slurry were applied per hectare (0 to 3500 pounds/acre) to a crop of orchardgrass. The manure was added below the soil surface (36). An application rate of 785 kg/ha (700 lbs/acre) manure nitrogen was excessive. Lesser rates were suggested.

Dairy manure application rates up to about 6725 kg N/ha, (6000 lbs N/acre) did not adversely affect corn yield (37). Nitrates, chlorides, and other salts moved through the soil profile at all application levels. The concentration of nitrate nitrogen below the soil surface was in excess of 10 mg/l, however crop uptake appeared to control the leaching at dairy manure application rates up to 94 metric tons/ha (42 tons/acre). Leaching occurred primarily before the crop was established and after it was removed. There appeared to be no economic advantage of manure applications much in excess of 45 metric tons/ha (20 tons/acre).

Investigations in Alberta (38) indicated that animal manure applied to soil at a rate of 70 metric tons/ha/yr (31 tons/acre/yr) for 40 years did not cause an undesirable buildup of nitrogen, phosphorus, or soluble salts in the soil.

Applications of settled wastewater from a swine production facility using a high rate flood application approach was not satisfactory (39) on silt loam. Application rates

of 2.5 to 10 centimeters (1 to 4 inches) were used. Percolates collected up to 75 centimeters (30 inches) below the surface had high COD (300-1010 mg/l), ammonium nitrogen (120-288 mg/l), and total phosphate (105-330 mg/l) concentrations. It was concluded that either a greater degree of treatment was required before land disposal or that the application rate must be reduced.

Large applications of cattle wastes have depressed the yield of crops, primarily by the accumulation of soluble salts (40). Increased accumulations of nitrate occurred at beef cattle manure application rates exceeding 740 metric tons/ha (330 tons/acre). The nitrates did not leach appreciably below a soil depth of one meter (3.28 feet). The study was done in Kansas which has a different precipitation pattern than that of the Great Lake states. Leaching in the more humid Great Lake states may be expected to be greater than that noted.

The application of dairy manure to land was studied in California (41). Spreading the manure from ten dairy cows on 0.4 ha (1.0 acre) resulted in excessive nitrate movement to the groundwater. The manure from 7.5 dairy cows spread on one hectare (3 cows/acre) was suggested as a rate that would keep the nitrate nitrogen concentrations at less than 10 mg/l in soil water below the root zone. This rate was estimated to contribute about 246 kg N/ha (220 pounds/acre) to the soil.

The amount of manure nitrogen that can be applied to soils in Ontario, Canada without causing excess nitrogen in the soil water has been estimated at about 350 kg/ha (310 lbs/acre) (42).

English investigators stress that the application of manure nitrogen must be correlated to the type of crop that is being grown (43). With intensive grass production, the nitrogen application rate was suggested as high as about 560 kg/ha/yr (500 lbs/acre/yr) whereas if barley were being grown, the nitrogen application rate may have to be as low as 34 kg N/ha/yr (30 pounds/acre/yr) because of the lower utilization of nitrogen by the barley plant.

Of the applicable regulations covering the land disposal of animal manures, only the Maine Guidelines (44) specifically state manure application rates which are based on the nitrogen content of the manure. Annual manure nitrogen applications are limited to 673 kg/ha (600 lbs/acre) where a crop is removed annually, and 560 kg/ha (500 lbs/acre) where no crop is removed. Many Maine soils are limited

to rates lower than these maximums because of less optimum characteristics. The suggested land area necessary to dispose of animal wastes in Ontario (45), South Carolina (46), and New York (47) are based on nitrogen application rates. The South Carolina guidelines assume a 50% nitrogen loss between excretion by an animal and application of the wastes to the land. The New York Guidelines suggest manure application rates as a function type of crop, land slope, time of application, and general soil drainage characteristics. The Guidelines recognize that manure application rates should be less on sloping land, poorly drained land, and in the winter.

The nitrogen application rate is the key operational parameter as far as determination of the adequate land disposal areas is concerned. However it is not the nitrogen application rates per se which are of concern. Rather it is the interrelationship between the rate of nitrogen applied, the rate of nitrate production in the soil, the rate of nitrate utilization by a crop, the presence of excess water for leaching, and the rates of ammonia volatilization and denitrification that are fundamental to the determination of permissable manure nitrogen application rates.

Manure application limitations based on its phosphorus content have been considered. However phosphorus is immobilized in the surface layers of the soil and is not leached from the soil in significant quantities. The ability of a soil to fix phosphorus is not infinite. Phosphorus can penetrate sandy soils and soils with high porosity and contribute to the phosphorus in tile drainage. It is unlikely, however, that phosphorus contamination would be a problem under application rates which are selected to avoid nitrogen contamination of groundwater.

4.5.2 Runoff

When manure is applied to a field, there is an opportunity for the movement of manure and nutrients into receiving bodies of water via runoff. The risk can be greater when manure is spread on frozen soil.

When animal manures were spread on frozen soil, an average of 10% of the nitrogen, 6% of the phosphorus, and 33% of the potassium was lost under conditions of an early spring runoff (14). When manure was spread in the summer and incorporated into the soil, runoff losses of nutrients

were less than on unmanured ground. The manure had a beneficial effect on reducing runoff by increasing water infiltration. Similar results have been reported by other investigators (48).

The application of manures to frozen ground does not always result in stream pollution. When 202 metric tons of dairy manure were applied per hectare (90 tons/acre) immediately prior to a heavy snow, the nitrogen and phosphorus content of the ultimate runoff was less than that in the runoff from a plot which received 100 metric tons/ha (45 tons/acre) on top of a layer of snow just preceding runoff (49). Manure application in the winter may not be a significant cause of contamination via runoff if the soil is not frozen, if the manure is incorporated into the soil before precipitation of snow melt causes runoff, and if the manure is not applied to extremely sloping land.

Control of the problem of manure disposal on frozen ground does not necessarily lie in a complete restriction of such disposal during the winter months. Rather, guidelines should be set up that identify the timing of such disposal in relation to rain and snow occurrance, the likelihood of runoff from the land, and the proportion of the time the soil is frozen. Guidelines should be developed on the basis of the probability of runoff from manured lands whether they are frozen or not. A winter spreading regulation which allows for the spreading of manure on frozen ground with the possibility of runoff every 10 years, might be acceptable. Such a runoff probability is consistent with those in the EPA effluent guidelines for feedlots. These guidelines are described in detail in Section 5.

Data indicating the contaminant load from land used for manure disposal is sparse. Available information from two detailed studies are noted in Table 14.

The water pollution significance of feedlot or barnlot runoff or runoff from lands receiving manure depends upon whether the runoff actually enters surface waters. Most of the available data represents information on the characteristics of runoff leaving a plot of ground or a feedlot. These characteristics do not necessarily represent what finally gets into a stream. Fields isolated from drainage ways by vegetated land of little slope may be of no practical concern as a runoff pollution source.

Several states have proposed controls on spreading manure near water capable of leaving the farmer's property.

TABLE 14
CHARACTERISTICS OF RUNOFF FROM PASTURE LAND AND LAND USED FOR MANURE DISPOSAL

	Constituent							
	BOD	COD	TOC	NO_3-N	Total N	Total P	Remarks	Ref.
North Carolina (g/ha/day)+	30	---	63	---	7.6	2.8	-mixed grains and orchard, swine waste spread	50
	17	---	124	1.8	10.4	0.6	-pasture, corn, orchard, swine waste spread	
	26	320	72	8	10	1.8	-poultry waste spread on 5 acres three times a year	
	1850	4350	2250	14	435	---	-22 tons of poultry waste spread on 4 acres once	
Wisconsin (kg/ha/yr)++								
-manure not spread	---	---	---	---	4.4 ** (3.6-5.5)	1.3 (1.2-1.5)	-three year average-1967-69 dairy cattle manure spread at the rate of 15 ton/acre	51
-manure applied*								
-winter	---	---	---	---	12.7 (3.0-27)	2.9 (1.0-5.8)		
-spring	---	---	---	---	3.8 (3.0-5.2)	0.8 (0.7-1.0)		

*High values were due to a thaw and a 0.75" rain immediately after spreading manure in winter; manure spread in the spring was incorporated into the soil after spreading.

**Values in parentheses indicate reported range of values.

+1.0 g/ha/day = 8.9×10^{-4} lb/ac/day

++1.0 kg/ha/yr = 0.89 lb/ac/yr

Maine guidelines require that no manure be spread within 7.6 meters (25 ft) of the outer edge of the normal high water mark of water courses. A distance of 30.5 meters (100 ft) between spreading areas and wells, springs, or lakes also is required (44). Iowa laws indicate that animal confinement areas which are more than a certain distance from a water course need not register or obtain permission for the operation of the facility (52). The distance depends upon the type of animal. For dairy cattle it is 92 meters (300 feet) per 100 animals, for beef cattle - 61 meters (200 feet) per 100 animals, for feeder pigs - 3 meters (10 feet) per 100 animals, and for broiler and layer chickens - 0.3 meters (1 foot) per 100 birds. The implication of such requirements is that these distances are adequate to avoid water pollution from the animal operations.

Rationale for establishing such minimum distances is not well documented. The concept, however, is sensible. A well vegetated strip of land between the site of manure application and the surface water will decrease the contaminants in land runoff considerably. A close growing crop such as grass will have a far greater filtering efficiency than a cultivated crop such as corn. Greater widths should probably be left unmanured if the field is cultivated to the edge of the water course. The degree to which pollution control grass strips are accepted by pollution control agencies is unknown.

4.6 Bacterial Contamination

Information on the effect manure application rates have on pathogen survival also is sparse. In general, application rates and conditions which favor runoff losses of manure also present the risk of pathogens being carried to surface waters.

The list of infectious disease organisms common to man and livestock is lengthy, including a number that can be waterborne. When drainage or runoff from animal production units reaches a water course, a potential chain for the spread of disease has been initiated. Salmonella organisms have been isolated from animal fecal specimens, runoff from animal confinement operations, and from waterholes from which animals drank (53, 54). Two organisms, S. dublin and S. typhimurium were the salmonella organisms most commonly found in the contaminated water.

A detailed study of microorganism survival in manure applied to soil indicated that frequent recontamination of

soil with dairy manure slurry resulted in a modified soil environment which favored extended survival of Salmonella enteritidis (55). The soil was a Scranton fine sand. The study suggested that pathogen survival in lands treated with manure is a problem which should be considered and controlled when contemplating the design and operation of a waste treatment process involving sprinkler irrigation on porous soils.

A study of the effectiveness of soil to remove bacteria in animal wastes indicated that coliform and entercoccus bacteria were removed by adsorption during soil percolation and by die-off because of the inability to compete against the established soil microflora (56). Greater than 98 percent removal occurred in 0.36 meters (14 inches) of soil.

Large increases in fecal bacteria occurred in tile drain effluent from a sandy loam soil on which swine slurry had been spread (57). Soil permeability is the property having the greatest influence on whether a soil will be a pollution risk when used for waste disposal. A soil with very high permeability is likely to allow the movement of wastewater into the ground at a rate such that bacterial removal may be inadequate. Worm holes and cracks in the soil can act as conduits for the wastewater and may be the cause of the bacterial and contaminant concentrations found in tile drain effluent and lysimeter samples.

The disease potential inherent in the disposal of animal wastes on land is unknown but is considered not to be critical. No significant outbreaks of disease have been traced to bacteria in manure and runoff from land used for manure disposal. However, the possible presence of human and animal pathogens in runoff and effluent from livestock waste treatment systems suggests that caution should be exercised in using such water for body contact sports or for a water supply. Practices to control the runoff will minimize discharge of the bacteria as well as organic and inorganic contaminants from these operations.

4.7 Livestock Grazing

The potential water quality problem associated with this livestock production process has drawn little attention from water pollution control agencies or researchers. Consequently, a minimum of information is available to determine the magnitude of the problem from this activity. When on range and pasture systems, the livestock are dis-

persed over a large land area and natural waste degradation will occur. The water pollution potential of range or pasture livestock production should be of much less concern than that of confinement livestock production. In addition, crop, forest, or brush land may exist between the pasture and surface streams. Such land will help reduce any contaminants in pasture land runoff to near "natural" levels. Where animals have direct access to streams, animal urine and feces can be discharged directly to such waters.

Some information is available which permits an estimate of the magnitude of the problem and the quality of runoff from range and pasture land. Studies in North Carolina noted (50) that a beef herd grazing on the banks of a stream resulted in larger losses of total nitrogen, phosphorus and organic carbon to a stream per animal than did manure spread adjacent to another stream. Loads in the runoff from a variety of pastures are listed in Table 15.

Data obtained in this study (50) illustrated the relative pollutional effects of pastured animals. The study noted that: a) direct access of animal waste to surface waters should be prohibited; b) points of animal concentration should be located away from streams; and c) vegetation should be provided between areas of animal concentration and drainage paths or surface waters to intercept any contaminant.

Runoff from range land that had low intensity agriculture and no evidence of chemical fertilizer uses contained 0.56 kg NO_3-N, 0.067 kg total P, and 0.02 kg soluble P per hectare per year (0.5, 0.06, and 0.19 pound/acre/yr respectively). Seventy percent of the nitrate-nitrogen and total phosphorus entered the lake under study during the period from February to May (58). Precipitation contributed over 70% of the nitrogen to this lake.

A study of the bacterial indicators in a partially grazed Vermont watershed (59) noted that both total and fecal coliforms organisms in storm runoff from the watershed were significantly higher than those of the usual non-storm values. Bacteria concentrations patterns paralleled storm hydrograph patterns. Data on other pollutional parameters were not obtained in the study. Total coliform concentration in the runoff from hayfields were similar to those in runoff from the partially grazed area. Fecal coliform concentrations were greater in the grazed watershed runoff than in the hayfield runoff. Total coliform concentrations were as high as 80,000 and fecal coliforms were as high as 14,000 bacteria per 100 ml in the partially grazed

TABLE 15
CHARACTERISTICS OF RUNOFF FROM PASTURE LAND (50)

LAND USAGE	AVERAGE CONSTITUENT (g/ha/day)* BOD	COD	TOC	NO_3-N	TOTAL N	TOTAL P
250 HOG DRYLOT, ROW CROPS, WOOD, GRASSLAND	35	--	97	1.7	7.7	3.5
PASTURE FOR 50-100 DAIRY COWS PLUS CORN	44	750	150	4.0	38	25
PASTURE FOR 160 COWS ON 15 ACRES	3450	10,700	3750	47	400	130
35 BEEF COWS ON 15 ACRES OF PASTURE	46	720	150	--	5.8	1.8

*1.0 g/ha/day = 8.9×10^{-4} lb/ac/day

watershed runoff.

The water quality from grazed and ungrazed watersheds has been compared (60). An unknown number of sheep and cattle grazed the areas under study. Runoff from the grazed areas had higher total coliform, fecal coliform, and fecal streptococcus organism concentrations than did runoff below the non-grazed areas. Physical and chemical parameters were measured and are noted with the bacterial concentration in Table 16. Nitrogen and phosphate concentrations were low. Although there were slight differences between the grazing areas, the values were not significantly different from values before or after grazing. Thus livestock grazing had no apparent effects on the chemical and physical values measured in the study. The increased bacterial concentrations measured in the study did not persist after the runoff mixed with a larger stream. Flow was not measured and area yield rates could not be determined.

4.8 Current Research

Based upon a review of Notices of Research Projects (N.R.P.'s) available from the Smithsonian Science Information Exchange, there were 109 research projects that were active in 1971-73 and that dealt with some aspect of potential pollution of surface and groundwaters from livestock production units. These projects were supported by the Office of Water Resources Research, U.S. Department of Interior; Environmental Protection Agency; Cooperative State Research Service, U.S. Department of Agriculture; Soil and Water Conservation Research Division, Agricultural Research Service, USDA; Agricultural Engineering Research Division, ARS-USDA; Sea Grant Office, National Oceanic and Atmospheric Administration; various state governments; the Australian Government; and the Rockefeller Foundation.

The research projects were of a nature to provide information on:

- general nutrient balances to lakes including from land used for livestock production
- land disposal of animal wastes with emphasis on both the immediate and long term effects and including information on organic, inorganic, and bacterial constituents of soil, soil water, and surface runoff

TABLE 16
COMPARISON OF MEAN VALUES-RUNOFF FROM
GRAZED AND NON-GRAZED AREAS (60)

station Parameter	A grazed	B grazed	C ungrazed
Nitrate (mg/l as NO_3)	0.14	0.22	0.15
Phosphate (mg/l PO_4)	0.25	0.16	0.16
Turbidity (JTU)	2.6	1.2	0.23
Total Coliform (counts/100 ml)	240	107	18
Fecal Coliform (counts/100 ml)	88	38	4
Fecal Streptococci (counts/100 ml)	325	101	27
Total Heterotrophic bacteria (counts/100 ml)	3780	2780	3990
pH	8.4	8.3	8.3
Number of Samples Analyzed	38	35	31

- feedlot runoff characteristics, precipitation relationships, and management approaches

- runoff from pastures

The majority of the research was being conducted at Universities and USDA field stations. A number of studies were being done at fully instrumented watersheds in many states. The investigators, the objectives of the research, and the available facilities appeared quite adequate to better understand the magnitude of contaminants from livestock operations and methods to manage the wastes to avoid water pollution. Few of the NRP's provided information on the progress of the research and thus no evaluation of the project results could be obtained.

Only one NRP indicated that the information would be used to characterize the runoff losses on a per unit area or per unit of runoff basis. A few of the NRP's indicated that the project results would be used to develop feasible land waste application rates which would avoid surface or groundwater pollution problems. Many of the projects appeared to obtain sufficient information to determine unit runoff yield rates as related to waste application rates or waste management conditions, however, it was not clear that the results would be interpretated in such a manner.

Information was generally collected in terms of runoff, soil, and soil water characteristics from a specific plot of ground or livestock operation. Many projects appeared to look at a specific type of contaminant such as bacteria, nitrogen, oxygen demanding material, or toxic elements. Few of the projects appeared comprehensive enough to obtain information on more than a limited number of pollutant characteristics. This is unfortunate since with a minimum of additional effort, comprehensive data could be obtained which would permit a more complete understanding of the potential pollution problems and permit presentation of the results in terms of runoff yield rates and manure application rates.

The lack of comprehensiveness of many projects probably resulted from inadequate funding and from the lack of an interdisciplinary nature of most of the projects. The vast majority of the projects appeared to be the result of individuals within a single discipline.

None of the projects appeared to evaluate the changes in runoff characteristics that occur during transport across land and in creeks and streams. As a result, while many of

the projects appeared to provide information on the runoff emanating from livestock facilities and manure disposal operations, none appeared to consider the fate of the pollutants during subsequent transport or the actual impact on surface waters.

4.9 Summary

Although the available information on the pollution of surface and groundwaters from livestock operations is not extensive and in many areas is sparse, there is adequate information to compare the potential sources, to identify the relative need for control, and to indicate additional studies.

Runoff from intensive animal feedlots is the most severe problem because of its large pollutional load since the animals are more dispersed. However, wherever livestock are concentrated to the point that vegetation will not grow, such as in barnlots, cattle loafing areas, and around feeding areas, the runoff can contain significant pollutants and also should be prohibited from entering surface waters.

When animals are on pasture or grazing land, the runoff may contain elevated bacterial concentrations but no significant concentrations of other pollutants. Very little information exists on the water pollution potential from these land areas. Because such land areas are larger than those for confinement or intensive livestock operations, a detailed quantitative study conducted in the Great Lakes states could be useful to clearly identify the contribution of such land areas to the pollution of the waters in the Basin.

Seepage from stacked or stored manure is of low volume but has high pollutant concentrations. Such seepage should not be permitted to enter surface waters.

If livestock manure is improperly applied to land, water pollution problems can occur. Good practice guidelines such as reasonable application rates, distance from streams, and timing of applications, are available and can minimize pollution problems from manure disposal on the land. However, considerably more information is needed to relate manure disposal management to water pollution potential. Definitive data is not available to relate the quantity of pollutants in runoff or soil water to manure disposal rate,

timing, or types of crops. Additional studies in this area would be very beneficial. Such studies also would shed light on the appropriate land disposal rates of any organic waste and thus would be helpful in estimating application rates for sewage, sewage sludge, and organic industrial wastes.

Bacterial contamination from livestock operations or manure disposal does not appear to be a serious problem.

Groundwater pollution problems also do not appear to be serious. However very little information is available to identify actual conditions or trends.

Although numerous studies designed to document the pollution problems from livestock operations have and will continue to be conducted, very few have been comprehensive enough to clearly identify and quantify the pollution from the average, less intensive, livestock production operations of the type that exist in the Great Lake states. Two or more detailed studies, conducted at sites in Michigan and New York since largest number of animals are in these states, would be useful to provide the necessary detailed knowledge. These studies should be on watersheds that have different types of livestock and manure management operations. Data on physical, chemical, bacteriological, and hydrological characteristics of the runoff should be collected. The runoff data should be quantified in terms of pollutant load per unit land area, and per unit of precipitation wherever possible. Evaluation of pollutant changes during transport across land and in streams should be an important part of such studies. Individuals from a number of disciplines, such as agronomists, soil scientists, aquatic biologists, farm managers, and economists, should be involved. It will be necessary to conduct the studies over at least three and preferably more years to obtain the breadth of information that is necessary to identify the magnitude of the problem and better management approaches.

Section 5

REGULATIONS

5.1 General

Both federal and state regulations can apply to the control of water pollution resulting from agricultural operations. Available legislation generally refers to pollution irrespective of its source although specific types of pollution have been identified in such legislation.

It is accepted that surface and groundwaters receive contaminants from many natural situations that are not controllable by man, are not caused by man, and would exist in the absence of man. Examples of such "natural" or "background" pollution have included runoff from rural and forest lands, swamps, decaying vegetation such as leaves and crop residue, and the wastes from wild animals. Such pollution is difficult to control because of its diverse nature, lack of controllable point sources, and inadequate knowledge concerning feasible abatement techniques. It has been assumed that this type of pollution is of small significance compared to other sources of pollution.

Recent developments in agricultural production techniques have altered the traditional concept of considering pollution from agricultural production as "background" or uncontrollable pollution. As a result, greater attention is being focused on this source and available legal restraints are being applied increasingly to agricultural operations.

Most available regulations are of little value in the control of the downward movement of contaminants through the soil and into the groundwater. Proof of the source of the contaminant and of causative negligence is difficult to establish for most of the materials commonly involved in agricultural operations. To date there has been little agreement on the details of the control that may be suitable to avoid groundwater contamination.

5.2 Federal

Federal pollution control legislation has been developed since the turn of the century but with increasing rapidity since 1948. The basic policy and philosophy of water pollu-

tion control in the United States can be found in the Water Pollution Control Act of 1948 and subsequent legislation. The basic policy of federal water pollution control legislation is that: a) Congress has the authority to exercise control of pollution in the waterways of the nation, b) both health and welfare benefit by the prevention and control of water pollution, c) a national policy for the prevention, control, and abatement of water pollution shall be established and implemented.

The Water Quality Act of 1965 caused the federal government to assume a larger role in the control of water pollution. The 1965 Act included the provision for establishing water quality standards. Each state was to develop water quality criteria and a plan for implementation and enforcement. The water quality criteria adopted by a state were to be the water quality standards applicable to the interstate waters or portions thereof within that state.

A change in the water pollution control philosophy of the nation occurred with the establishment of these standards. The emphasis is now on the amount of wastes that can be kept out of the water rather than on the amount of wastes that can be accepted by the waters without causing serious pollution problems. This philosophy will guide acceptable waste treatment and disposal methods as well as legal actions in the future.

An unmistakable implication of federal and state water pollution control regulations is that facilities producing agricultural wastes will have to consider the impact of existing legal restraints in all expansion plans and in the establishment of new facilities. The facilities should be located where treatment and disposal can be obtained at minimum cost and without resulting in water pollution.

To integrate pollution control activities at the federal level, the Environmental Protection Agency was established in 1970. EPA contains the federal activities dealing with solid waste management, air and water pollution control, and water hygiene among others and has the responsibility to coordinate such environmental quality activities to minimize pollution transferral.

The 1972 Amendments to the Federal Water Pollution Act (PL92-500) continued the emphasis on keeping wastes out of surface waters. The Amendments declared that it is the national goal that the discharge of pollutants into the navigable waters of the United States be eliminated by 1985. To achieve this goal, effluent limitations on point waste

sources were to be achieved by the application of best practicable control technology by 1977 and of best available technology by 1983. The burden will be on industry to show that their treatment facilities represent the maximum use of technology within its economic capability. A number of agricultural operations such as dairy product processing, fruit, vegetable, and seafood processing, feedlots, and fertilizer manufacturing are included among the point sources that will be subject to effluent limitations.

With the passage of the 1972 Amendments, national pollution control emphasis shifted from water quality standards to effluent limitations. This approach was taken because of difficulties in linking waste discharge quality with stream quality and in enforcing previous legislation. The basic assumption, constrained only by the availability of economical control technology, is that the nation will strive toward complete elimination of water pollution.

The 1972 Amendments did not fully extend its provisions for surface water quality control to groundwater but they do contain evidence of a greater concern for groundwater quality. The Amendments provide for the development of comprehensive programs for preventing, reducing, or eliminating the pollution of navigable waters and groundwaters and improving the sanitary condition of surface and underground waters (Section 102a).

Agriculturally caused pollution problems officially were recognized for the first time in the 1972 Amendments. Certain agricultural waste point sources were subject to effluent limitations, specific appropriation authorization for agricultural waste research and demonstration projects was included, and area and statewide wastewater treatment management plans were to include a process to identify agriculturally related non-point sources of pollution and to set forth procedures to control such sources. The EPA Administrator was authorized to issue guidelines for identification and control of pollution from a variety of non-point sources, including those from field, crop, and forest land.

The concept of recycling potential municipal sewage through production of agriculture, silviculture or aquaculture was encouraged. This latter concept will require close management of such recycle to minimize subsequent water pollution and health problems. The concept will increase the overlap between agricultural and urban activities and their relation to pollution problems. Transferral of pollution from one location (urban) to another (rural) will

not be successful or tolerated.

The National Pollution Discharge Elimination System (NPDES) was a part of the 1972 Amendments. Under this legislation EPA is to establish a permit system for discharges of pollutants from point sources into navigable waters of the United States. The effect of this legislation potentially is to require a permit for every significant point source of pollution in the country including those from animal feedlots and other animal production operations. However EPA regulations issued in July 1973 excluded smaller farms from NPDES coverage unless they are considered as a significant contributor to water pollution.

On July 5, 1973, EPA published rules and regulations aimed at acquiring information on point sources of pollution resulting from agricultural and silvicultural activities (61). These regulations identified the size of operations that are required to provide the information and the type of information required. In addition, the regulations defined the term "animal confinement facility". This definition helps define the difference between intensive animal feedlots and more broad based animal production.

The term "animal confinement facility" was defined as:

> "A lot or facility used or capable of being used for the feeding or holding of animals (other than fish or other aquatic animals), but does not include land used for the growing of crops or vegetation for animal feed. Two or more animal confinement facilities under common ownership are deemed to be a single animal confinement facility if they are adjacent to each other or if they utilize a common area or system for the disposal of wastes."

The animal production facilities which were required to obtain a permit from state or interstate programs under NPDES were the following:

Animal confinement facilities if they contain or at any time during the previous 12 months contained, for a total of 30 days or more, any of the following types of animals at or in excess of the number listed for each type of animal:

(i) 1,000 slaughter and feeder cattle;

(ii) 700 mature dairy cattle (whether milkers or dry cows);

(iii) 2,500 swine weighing over 55 pounds;

(iv) 10,000 sheep;

(v) 55,000 turkeys;

(vi) If the animal confinement facility has continuous overflow watering, 100,000 laying hens and broilers;

(vii) 5,000 ducks.

Facilities which produced more than one type of animal also were required to obtain a permit if they contain, or at any time during the previous 12 months contained, for a total of 30 days or more, a combination of animals such that the sum of the following numbers is 1,000 or greater: the number of slaughter and feeder cattle multiplied by 1.0, plus the number of mature dairy cattle multiplied by 1.4, plus the number of swine weighing over 55 pounds multiplied by 0.4, plus the number of sheep multiplied by 0.1. In addition, any agricultural or silvicultural activity which has been identified by the Regional Administrator or the Director of the State water pollution control agency or interstate agency as a significant contributor of pollution, also must apply for a permit. The intent of this statement was to assure that animal production facilities which contained less than the above number of animals but which have or are shown to contribute to water pollution also would be identified, be required to submit data, be required to obtain a permit, and thus be under scrutiny and control.

A numerical cutoff level is necessary in any set of regulations. Levels that are set will tend to represent best judgement based upon experience and the desire to effectively administer the regulations. As more experience is gained with the cutoff levels and administration of the regulations, it is possible that the cutoff levels will be lowered to bring more feedlots under scrutiny. However, some cutoff level based upon size of feedlot is desirable. To have no numerical cutoff would overwhelm the agencies with applications and make the regulations ineffective.

The cutoff levels in the July 5 regulations were set at levels that were felt adequate to control the largest discharges first and in doing so, cover the worst potential sources of pollution. The levels also excluded the smaller operations on the theory that the expenditure of resources necessary by EPA to include them would be disproportionate to the potential environmental improvement. In addition,

EPA and the states have inadequate personnel to process the applications that could occur if every livestock producer were required to obtain a permit.

Although estimates vary, there are an extremely large number of livestock operations in the nation. In the Great Lakes Basin, there are about 130,000 livestock operations with sales of at least $2500 per year. If all of these were required to obtain a permit, it will be an enormous task simply to process these applications let alone attempt to enforce the guidelines on specific farms. To implement a regulatory program that would require surveillance of all the "feedlots" in the Great Lakes Basin as well as the nation would be an awesome and perhaps impossible task.

For the nation as a whole, the cost of writing permits for every feedlot was estimated to be in excess of $1 billion (62). In view of the expense of time needed to evaluate and process the applications, it is likely that the states and EPA will continue to utilize the basic concepts in the July 5 regulations to minimize pollution from feedlots.

Irrespective of a cutoff level that may exclude certain size feedlots from obtaining permits, it is difficult to see how any cutoff level will exempt feedlots which present a pollution problem from being regulated. The larger feedlots will be identified by any cutoff levels and smaller ones which have or do cause a pollution problem which will be identified by the regional or state agencies. The public is pollution control conscious. Hunters and fishermen are conservation and environmentally oriented as are vacationers and others who use rural land for recreation. These individuals frequently bring environmental problems to the attention of state and local agencies. Even where federal and state agencies lack the personnel to seek potential polluters, the above individuals are likely to identify and report environmental problems.

These rules and regulations were aimed at point sources of pollution due to agricultural and silvicultural activity. The rules and regulations excluded discharges such as irrigation return flow and runoff from orchards, cultivated crops, pastures, rangelands, and forest lands, thereby indicating that such discharges are to be considered as non-point sources of potential pollution.

The application for a permit to discharge point sources of waste from agriculture (Short Form B) under NPDES included the following required information: name and location of the facility, ownership, previous notice of complaints,

directions to the facility, map or photograph identifying overall dimensions of facility, surface drainage, waterways, areas of manure disposal, type and size of animals, and expansion possibilities.

Both the U.S. Department of Agriculture (4) and the Bureau of Sport Fisheries and Wildlife of the Department of Interior (63) have indicated that the EPA regulations of July 5, 1973 appear realistic.

On September 7, 1973, EPA published proposed effluent guidelines for existing sources and standards of performance and pretreatment standards for new sources for feedlots (64). The term "feedlot" was defined as a

> "concentrated, confined animal or poultry growing operation for meat, milk or egg production, or stabling, in pens or houses wherein the animals or poultry are fed at the place of confinement and crop production is not sustained in the area of confinement."

Such a definition is a reasonable definition of an intensive animal feedlot as used in this report.

The proposed effluent guidelines were applicable to beef cattle, dairy cattle, swine, sheep, horses, broiler chickens, laying chickens, turkeys, and ducks. For all animal categories except ducks, the effluent guidelines representing the application of best practicable control technology currently available was proposed as

> "no discharge of process wastewater pollutants to navigable waters, except for runoff which is not contained by facilities, designed, constructed, and operated to contain all process wastewater in addition to the runoff from the 10 year, 24 hour rainfall event as established by the U.S. Weather Bureau for the region in which the point source discharger is located."

The guidelines representing best available technology economically available was essentially that as noted for best practical control technology except that runoff control facilities were required to be designed, constructed, and operated to contain all process wastewater in addition to the 25 year, 24 hour rainfall event as established by the U.S. Weather Bureau for the region in which the point source discharger is located.

Standards of performance for new sources were proposed to be the same as for that for best available technology economically available.

These effluent guidelines establish the "no discharge" concept as the best available technology for these feedlot categories. The guidelines were established based on the assumption that the land would be the ultimate receptor of these animal wastes and that land runoff was a non-point rather than a point source problem.

The duck category had somewhat different proposed effluent guidelines. Point source discharges were permitted under the use of best practicable control technology currently available. The effluent limitation representing the degree of effluent reduction obtainable by the application of such technology was indicated as:

> BOD_5 - Maximum for any one day: 1.66 kg (3.66 lb^5) per 1,000 ducks and maximum average of daily values for any period of 30 consecutive days: 0.91 kg (2.00 lb) per 1,000 ducks.
>
> Fecal coliform - Maximum at any time: 400 counts per 100 ml.

The guidelines representing best available technology and standards of performance for new sources were proposed to be the same as that established for the other animal sources, i.e., no point source discharge except for runoff not contained by facilities designed, constructed, and operated to contain all process wastewater in addition to the runoff from the 25 year, 24 hour rainfall event as established by the U.S. Weather Bureau for the region in which the point source discharger is located.

The Natural Resources Defense Council has instituted Court action to require EPA to develop effluent limitation guidelines for confined animal feeding facilities whose sizes were smaller than those identified in the July 5, 1973 regulations. As this report is written, the action has not been concluded.

Interested parties had the opportunity to provide comments on the proposed effluent guidelines. These comments were reviewed and analyzed as the final effluent guidelines were developed. The final effluent guidelines for the feedlot point source category were published in the Federal Register on February 14, 1974 and became effective on April 15, 1974 (65). Several changes from the proposed

effluent guidelines of September 7 were made.

The definition of a "feedlot" was modified slightly to show that if either crop or forage growth or production is sustained, the operation would not constitute a feedlot. The underlined words were added to the definition. Therefore, when conditions exist where the density of livestock in any given area precluded the growth or production of crops or forage, and the other elements of the definition were met, those conditions would describe a feedlot.

The importance of the modification lies in the interpretation that livestock operations which may constitute a feedlot in the strict sense, particularly smaller facilities, may consider management options which disperse or decentralize livestock in order to provide crop or forage growth. Under the latter circumstances the combined effect of soil and vegetative assimilation of manure and the lower rate of manure deposition per unit area could reasonably be expected to preclude any significant pollution problem. Livestock operations which cannot be defined as a feedlot by the modified definition presumably do not fall in the point source category.

These final regulations make it clear that exceptions to the "no discharge" standard are not limited to those occasioned by a given storm event, but include discharges attributable to the cumulative effect of rainfall or snowmelt over a period of time.

The regulations indicated that "EPA does not intend to establish operating criteria for feedlot waste control facilities. Rather, the regulation establishes a performance standard with an exception and, to qualify for this exception, the operator is free to choose any method of operation, providing the method of operation causes the facility to contain process generated wastewaters and usual rainfall events. Such methods may normally include removal of the stored wastewaters for purposes of irrigation or other land utilization of the waste in accordance with accepted agricultural practices which prevent discharge to navigable waters, and maintain the facility in a state of availability to contain runoff from recurring or subsequent rainfalls." The key phrase is to "prevent discharge" and thus is the goal of the feedlot operator for whatever mode of operation he chooses.

It also was made clear that the February 14 regulations are directed at the large operations as identified in the July 5, 1973 rules and regulations. Applications of the

guidelines to smaller operations is under review by EPA.

The final guidelines recognize that all possible data may not have been available in developing the final guidelines and as a result, limitations in the final guidelines may require adjustment for certain operations in this category. An individual discharger or other interested person may submit evidence to the Regional Administrator (or to the State, if the State has the authority to issue NPDES permits) that factors relating to the equipment or facilities involved, the process applied, or other such factors related to such discharger are fundamentally different from the factors considered in the establishment of the guidelines. On the basis of such evidence or other available information, the Regional Administrator (or the State) will make a written finding that such factors are or are not fundamentally different for that facility compared to those specified in the Development Document. If such fundamentally different factors are found to exist, the Regional Administrator or the State shall establish for the discharge effluent limitations in the NPDES permit either more or less stringent than the limitations established herein, to the extent dictated by such fundamentally different factors. Such limitations must be approved by the Administrator of the Environmental Protection Agency. The Administrator may approve or disapprove such limitations, specify other limitations, or initiate proceedings to revise these regulations.

The rainfall events noted in the regulations have been defined more broadly to mean an "event with a probable recurrence interval of once in ten years or twenty-five years, respectively, as defined by the National Weather Service in Technical Paper Number 40, "Rainfall Frequency Atlas of the United States", May 1961, and subsequent amendments, or equivalent regional or state rainfall probability information developed therefrom."

The "no discharge" concept for all feedlot categories and the effluent guidelines for the ducks category that were part of the proposed guidelines were not changed and were incorporated as part of the final guidelines.

The application of these regulations to livestock operations in the Great Lake Basin appears adequate to minimize gross runoff pollution problems from these operations.

5.3 State

Legal controls on water quality have been recognized as traditional state responsibilities. Considerable variation has existed among states in their activities. While most state legislation does not mention agricultural pollution sources specifically, they are not excluded especially those that are point sources. Because of the increased interest in agricultural pollution control, some states have developed regulations for such control.

The control of water pollution from animal feedlot operations has come under specific regulations in a number of states. A summary of state regulations that existed as of 1971 (66, 67) indicated that the regulations can be grouped into three broad areas: a) specific regulations concerning feedlot construction, b) control of feedlots through existing regulations such as water quality standards or public nuisance regulations, and c) no specific regulations were considered necessary or applicable. A model state statute for animal waste control is available (68). Livestock operation registration and/or permit provisions are the predominant provisions provided by specific state legislation and administrative codes. These provisions exist in most of the major beef feeding states such as Arizona, Kansas, Nebraska, Texas and Iowa.

The Kansas regulations were one of the first that were used for control of water pollution from feedlots and have been used as an example by other states. These regulations were adopted in 1965 and stipulate that the operation of existing and proposed confined feeding operations must register with the Kansas State Department of Health. Where a potential for water pollution exists, suitable water pollution control facilities must be constructed in accordance with plans and specifications approved by the Department of Health. Approved facilities normally include: a) diversion of runoff from non-feedlot areas, b) retention ponds for all wastewater and runoff contacting animal wastes, c) application of liquid waste to agricultural land, and d) application of solid wastes to agricultural land. An important aspect of the regulations is that if, in the judgment of the Department, a proposed or existing confined feeding operation does not constitute a water pollution problem, provision of water pollution control facilities will not be required.

The success of feedlot regulations of the above nature can be glimpsed by consideration of statements made by the Acting Associate Director of the Bureau of Sport Fisheries

and Wildlife, Department of Interior (63). In the years up to 1970, feedlot pollution on the Flint Hills National Wildlife Refuge in Kansas had caused pollution problems in the adjacent John Redmond Reservoir. Approximately 100,000 cattle are in the feedlots in the watershed above the Reservoir. Since 1970, although precipitation patterns have been similar to earlier years, no pollution problems have existed. It was in the later 1960's and early 1970's that the Kansas regulations were widely enforced. Similar reductions in feedlot pollution problems occurred at the Buffalo Lake National Wildlife Refuge in Texas when the Texas Water Quality Board enforced feedlot runoff control regulations.

Modifications of the Kansas regulations, the use of permits to assure adherence to federal, state, and local regulations, and voluntary registration have been considered and adopted for feedlots in other states. While procedures vary among the states, applicable regulations contain general provisions that a feedlot operator obtain a permit from the appropriate state agency, correct any pollution hazard that exists, and assure that the operation conforms to all applicable federal, state and local laws. Minimum pollution control approaches may be required such as runoff retention ponds, dikes, and distances from dwellings and wells. A map of the area indicating land use, streams, roads, houses, wells, and other salient features frequently is requested.

As of 1971, of the states bordering the Great Lakes only Minnesota had specific regulations for control of waste from livestock feedlots. The Minnesota regulations were developed in regard to solid waste control. The Minnesota Pollution Control Agency and the Soil Conservation Service were developing an agreement covering the planning and installation of animal waste control structures (69). The Illinois Pollution Control Board has proposed regulations governing the construction, location, and operation of animal feedlots and animal waste treatment works. Hearings on these regulations are scheduled for early 1974. The other states have considered existing pollution control regulations adequate to control pollution from animal feedlots.

States have the option to develop and administer their own discharge elimination permit system (SPDES). If the state program meets certain minimum criteria, it will be approved by EPA in lieu of NPDES. To secure EPA approval, at least those operations subject to NPDES must be included in SPDES. It is possible that the requirements for the

state permit programs can be more stringent than for the national program.

Most of current state water pollution control legislation has been written for municipal and traditional industrial wastes. Livestock wastes have different volume and concentration relationships than do municipal and industrial wastes. In addition, many agricultural wastes do not reach streams except as land runoff. Disposal on the land rather than to streams is common. Use of the land as an integral part of agricultural waste management is a concept that must be recognized in applying available regulations.

To assist the control of agricultural wastes, many states have suggested or prepared "Codes of Practice" or "Good Practice Guidelines" for the disposal of agricultural wastes and the expansion of existing or expansion of new livestock facilities. Examples of such codes or guidelines can be found in New York, Wisconsin, and Ontario, Canada. These codes are intended to serve as a basis of reasonable operation for agricultural operations, including waste management, without being too specific in design requirements.

Key items in the codes or guidelines include suggestions of:

- land area to dispose of wastes
- waste storage capacity
- distance to human dwellings
- criteria for satisfactory waste handling and treatment facilities
- odor control
- solid waste disposal
- runoff control

The Agricultural Code of Practice for Ontario, Canada (45) incorporates many desirable concepts to minimize agricultural water pollution problems. Recommended land areas for waste disposal are included and were based on the number of equivalent animal units contributing the wastes and the characteristics of available soil. Minimum tillable land requirements were suggested for both loam and sandy soils. The minimum acreage was considered that necessary to avoid the risk of groundwater pollution by nitrogen compounds.

Suggestions for adequate distance between livestock buildings and human dwellings were included.

Approaches similar to those in the Ontario Code of Practice have been considered by other states. Guidelines for the state of Washington (70) include specific information on acceptable distances between land used for animal waste disposal and an open supply of water such as a river, creek, or lake. These guidelines also suggest minimum land areas which should be available for dairy manure waste disposal. Such land areas are based on the estimated fertilizer content of dairy manure. A minimum land area of 0.3 hectares (0.73 acres)/dairy cow or 1240 dairy animal manure days/hectare/year (500 days/acre/yr) was recommended for western Washington.

The Maine guidelines (44) outlined rates for the spreading of manure on the land. The maximum rates were developed based on the physical and chemical characteristics of Maine soils and knowledge of the movement of manure liquids and residues on and through each soil type. The limiting factor in determining application rates was the pounds of nitrogen to be applied per acre. Tables were provided to indicate manure application rates on specific Maine soil types. Suggestions related to spreading on slopes, snow or frozen ground, distances from surface waters, and type of crop were included in the guidelines.

The New York guidelines (71) discussed farm sites and planning, handling and storage of manure, manure disposal, and disposal of dead animals. Criteria for the frequency of manure cleaning, handling, and management and for the ventilation of confined animal operations were discussed for beef and dairy cattle, veal calf, horse, laying hens, ducks, turkey, and swine operations.

Where guidelines are available, an agricultural producer would be in a vulnerable position in respect to pollution problems if he had not adhered to them. Such guidelines do not insure that pollution or nuisance problems will not result. The advantage of codes or guidelines is that they represent best available practice yet can be altered more simply than regulations when better procedures become known and proven.

Not all of the suggestions in the above and other "Guidelines", especially those requiring land areas for waste disposal, are applicable to other areas in North America. Precipitation patterns, weather conditions, crop production patterns, and soil characteristics are

among the factors that will cause the good practice guidelines to vary from state to state and within a state.

Codes or guidelines of the above type can, however, offer a valuable approach for management of potential pollution problems from livestock production and represent a useful model for other governmental organizations. Codes or guidelines are preferable to governmental regulations since there is the danger that regulations will be applied uniformily to producers despite the wide variety of measures needed to properly protect the environment and the wide differences between agricultural production operations.

5.4 Inadequacies and Possible Solutions

Examination of the range of federal and state regulations which can affect animal feedlots reveals several areas of possible inadequacy: a) lack of controls on non-point sources of pollution, b) general lack of regulations aimed specifically at agricultural sources of potential pollution and c) few regulations dealing with potential groundwater pollution problems. In the field of water pollution control, the major federal and state programs are aimed at point sources of surface water pollution.

The current EPA-NPDES regulations exempts most animal production operations from obtaining a permit and therefore from immediate scrutiny. The zero discharge EPA effluent guidelines now in effect for feedlots means that point source discharges from animal production operations will be uncommon. As a result, any water pollution problems from such operations will be of a diffuse, non-point source nature.

Detailed state or national regulations for controlling general agricultural runoff do not seem necessary. With the exception of feedlot runoff, agricultural runoff is ubiquitous and not amenable to conventional regulatory control. Adherence to concepts included in the guidelines noted in the Section 5.3, State, will minimize the pollution caused by agricultural land runoff. Specific state regulations covering construction and operation of animal feedlots could satisfactorily control animal feedlot runoff and meet the February 14, 1974 effluent guidelines.

It is not desirable to develop rigid regulations which would force an animal producer to adopt methods which are not optimal for his operation. There are many ways to

modify operational practices to achieve reduced water pollution. Producers should be encouraged to use any appropriate approach to achieve pollution control rather than to simply comply with an uniform set of conditions prescribed by a state pollution control agency. The national effluent guidelines are strictly effluent quality requirements. It is up to the discharger to determine his own means of compliance using any available sources of information. State regulations should contain the same flexibility.

Farms on which livestock are produced exist in an almost unlimited combination of size, type of soil, slope of land, proximity to residential areas and surface waters, and combination of income producing activities. There is no one waste management and runoff control system that will fit every farm. Livestock producers do not have the technical knowledge to design and install the needed systems. Technical assistance is available to help producers solve their waste management problems from the Soil Conservation Service, the Extension Service, and through a few other federal and state agencies. However the demand for technical assistance will exceed the supply especially as the producers attempt to comply with the effluent guidelines for the feedlot industry.

The personnel problems associated with a large scale regulatory effort for livestock wastes deserve critical evaluation. There are inadequate personnel available to design the runoff control and containment facilities for farms with pollution control problems let alone design other waste management approaches for livestock operations. Based on Soil Conservation Service experience in planning, designing, and installing waste management systems, approximately 23,000 man years of technical assistance would be required to meet the needs of operations with water pollution problems (72). Some 7,000 man years would be required on a yearly basis between 1974 and 1977 to provide the technical assistance necessary to design and install waste management systems to meet proposed effluent guidelines. These estimated personnel needs are in addition to those needed to process permit applications and assure compliance with the guidelines.

In fiscal year 1973, the Soil Conservation Service provided about 390 man years of technical assistance to design and install 3800 waste management systems (72). Thus based on just the yearly needs caused by the effluent guidelines, about 20 times the current personnel would be needed. Cooperative Extension agents also can help producers meet their waste management needs. However, within the personnel and financial resources currently available, no great increase in man power commitment to the design of livestock waste

management systems can be expected.

The chief federal water pollution control enforcement agency, EPA, has only five full-time positions assigned to feedlot pollution and these are in research, one in Washington and four in the EPA, Ada, Oklahoma research laboratory. Other individuals in EPA such as those in Effluent Guidelines, permit programs, enforcement, and non-point pollution problems also have part-time involvement with feedlot pollution. The aggregate personnel in EPA assigned to feedlot pollution is very small considering the need created by the feedlot effluent guidelines and the permit regulations.

Other federal agencies and state pollution control organizations are unable to expand to meet the increased demand caused by a livestock waste permit and enforcement program without new personnel and financial resources. Many agencies can barely sustain existing programs with present staff and budget. To add a new comprehensive enforcement program at this time without new resources could damage existing programs.

Any regulations developed to control pollution to animal wastes should: a) prohibit practices under conditions which make them undesirable, b) make it possible to easily identify operations with the potential to cause pollution problems, c) make compliance with the law desirable and attractive to reduce the need for enforcement, and d) have the producer responsible for proving compliance with the regulations.

If it is deemed necessary to have a comprehensive enforcement program to control pollution from livestock wastes, additional personnel and financial support will have to be provided at both the federal and state levels.

Section 6

RESEARCH, DEMONSTRATION, AND INFORMATIONAL PROGRAMS

6.1 General

Problems associated with animal wastes have been under investigation for over a decade. Initial investigations focused on the magnitude of the problems and subsequent investigations largely emphasized treatment and disposal possibilities which would minimize air and water pollution problems. In recent years the investigations dealing with treatment and disposal have continued but there has been increasing interest on evaluating an animal production operation as a total system, including the waste management aspects.

As a result of these investigations, a considerable body of knowledge exists on how to handle, treat, and dispose of animal wastes with minimum adverse environmental effects. The land has been considered to be the ultimate acceptor of treated or untreated animal wastes and treatment and disposal approaches have assumed that wastes could be disposed of properly on the land. The emergence of non-point source water pollution considerations has caused practices associated with land disposal of wastes to receive greater scrutiny.

The major animal waste research and demonstration programs are supported by the U.S. Department of Agriculture through its many components and through various types of support to universities. The Environmental Protection Agency and other federal agencies such as the Office of Water Resources Research, Department of Interior, also support research relevant to animal waste management and agricultural land runoff.

Details of specific research projects are discussed in other sections of this report, Sections 4 and 8. The purpose of this section is to identify the type and source of research, demonstration and informational programs that are available.

6.2 Environmental Protection Agency

In 1967, the Federal Water Pollution Control Administration, now the Environmental Protection Agency, (EPA), assigned the animal feedlot wastes (beef, swine, and dairy) research and development responsibilities to the Robert S. Kerr Environmental Research Laboratory at Ada, Oklahoma. Responsibilities for poultry waste are assigned to the Southeast Water Research Laboratory at Athens, Georgia. The goals of the EPA animal waste program are to define the animal waste problem and its real and potential pollutional effects, to determine technically deficient areas of control, and to conduct and stimulate research, development, and demonstration of animal waste pollution control technology.

Specific details of this program can be obtained from:

Agricultural Pollution Control Section
Non-Point Pollution Control Division
Office of Research and Development
Environmental Protection Agency
Washington, D.C. 20460

or

Agricultural Wastes Section
Robert S. Kerr Environmental Research Laboratory
Environmental Protection Agency
P.O. Box 1198
Ada, Oklahoma 74820

Several documents are available which identify the activities and priorities of the EPA animal waste program (73, 74). Several large scale demonstration projects are in progress for the treatment and disposal of dairy, beef, hog, and poultry wastes using a variety of treatment and management approaches. EPA interagency efforts have included support of a study on beef cattle waste management with ARS-USDA in Nebraska, and on modelling efforts of land runoff with the Soil Conservation Research Division of ARS-USDA at Watkinsville, Georgia.

The EPA research and development program has identified the following areas as priority research needs:

- reprocessing and reuse systems for animal and poultry waste

- demonstration of treatment/land disposal systems for animal and poultry wastes from small operations

- demonstration of measures to control ancillary pollutants such as odor, dust, antibiotics, etc.

- methods to control pollutants from new and abandoned feedlots and containment/treatment/land disposal of snowmelt runoff

- animal management concepts which minimize pollution source problems

- dissemination of research and demonstration results and waste management guidelines

A bibliography of livestock waste management research literature has been prepared under EPA auspices and is available (75).

The Office of Air and Water Programs, EPA, also is interested in available practices of animal waste utilization and/or disposal. A report of such practices was prepared for the Office (76). The report focused on potential reuse and recovery processes for animal wastes such as gas or oil recovery and refeeding to animals. Land spreading was noted as the ultimate destiny of nearly all the animal manure currently produced.

The Effluent Guidelines Division of EPA has the responsibility of providing guidelines for effluent limitations to be achieved by "point" sources of waste discharge into navigable waters and tributaries of the United States. In the development of such guidelines, a report was prepared to provide the most current and accurate data for the animal feedlot industry, an industry identified as part of the point source categories for which guidelines were necessary.

This report (77) identified the pollutant characteristics from animal feedlots and assayed the available control technology. Eight animal types were included in the study: beef cattle, dairy cattle, swine, sheep, chickens, turkeys, ducks and horses. The report provides an excellent summary of many treatment and disposal processes for these animal types. Information on the processes includes: description of the process, development status, reliability and applicability, and available costs. This report is recommended as a concise summary of available water pollution and runoff

control technology for animal wastes.

As part of the requirements of PL92-500, the Administration of EPA is required to issue information relating to practices to control pollution resulting from runoff from agricultural fields and croplands. Under contract with EPA, the Economic Research Service, USDA prepared a report providing general information on alternative control measures and cultural practices that may be useful to control pollutants from agricultural activities (5). The report notes that appropriate animal and land management practices which should be followed include: a) spreading acceptable rates of manure uniformly on land, b) applying feedlot runoff effluent on land as recommended for specific site conditions, c) maintaining an adequate land to livestock ratio on pastures, and d) locating feeders and waterers a reasonable distance from streams and water courses.

6.3 U.S. Department of Agriculture

Research to support the land and water resource programs of the Department of Agriculture is conducted by the Agricultural Research Service (ARS). Some funds have been available for livestock waste pollution control. Specific pollution control projects include those on: reuse of animal waste as feed, composting of swine wastes, poultry waste applied to poorly drained soils, managing animal wastes in feedlots, managing cattle wastes in cold climates, storage of dairy cattle wastes, nitrate pollution from disposal of dairy waste, loss of nutrients spread on frozen ground, soil fertility maintenance with animal wastes, the role of soil in farm waste management, and relating agricultural practices to water quality. These projects are conducted at state agricultural experiment stations and at ARS regional research stations. ARS has projects underway on the land disposal of sewage effluents and sewage sludge and on the composting of sewage sludge.

Further information in ARS research projects can be obtained from:

Dr. Robert G. Yeck, Staff Scientist
Soil, Water, and Air Sciences
National Program Staff
USDA-ARS
Beltsville, Maryland 20705

Another USDA agency which is involved in research is the Cooperative State Research Service (CSRS). CSRS administers the federal grant programs for agricultural and forestry research conducted at state agricultural experiment stations and at other state institutions. A considerable amount of research dealing with animal waste pollution control is supported by CSRS at universities. Summaries of projects supported by CSRS are available through the CRIS abstracts identified in Section 6.5.

6.4 National Science Foundation

Two projects supported by the Research applied to National Needs Program (RANN) of NSF are likely to produce results of interest to the control of nutrients from agricultural operations. These projects deal with nitrate in effluents from irrigated lands and the economic and ecological consequences of inorganic fertilizer use. NSF also is supporting research illustrating the production of methane from animal wastes by anaerobic digestion.

6.5 Universities

A vast number of diverse research projects dealing with animal waste treatment and disposal as well as runoff control from agricultural lands are under way at many universities, especially the land grant universities and colleges. A considerable portion of such research is supported by the U.S. Department of Agriculture through its various divisions and components.

The Current Research Information System (CRIS) of USDA can provide project abstracts on the animal waste and land runoff projects conducted at the land grant universities. These abstracts include project title, investigators, objectives, approach, and progress report and publications when available. Inquiries concerning availability of the CRIS reports should be made to:

Mr. J.R. Myers
Current Research Information System
Cooperative State Research Service
U.S. Department of Agriculture
Washington, D.C. 20250

The CRIS abstracts dated March 15, 1973 were reviewed for this project. Information on current research related to this project are noted in the technical sections of this report.

6.6 Informational Sources

Three major sources exist for information on animal waste management and land runoff: the CRIS abstracts noted above, the Water Resources Scientific Information Center (WRSIC), and the Smithsonian Science Information Exchange (SSIE). WRSIC provides abstracts of articles in many areas including those of interest to this project. Abstracts of articles selected for inclusion are prepared by individuals sufficiently familiar with the area to prepare accurate material. Abstracts are published every two weeks, a yearly summary is available, and compliations of abstracts on specific topics can be obtained. The bibliography on livestock waste management noted earlier (75) was compiled from abstracts which were forwarded to both WRSIC and EPA, Ada, Oklahoma.

Information on the WRSIC abstracts can be obtained from:

Water Resources Scientific Information Center
Office of Water Resources Research
U.S. Department of the Interior
Washington, D.C. 20240

The information from CRIS and SSIE is of a similar nature, abstracts of research projects. The CRIS abstracts provide more detail about results, progress, and publications. SSIE provides selected information packages consisting of Notices of Research Projects (NRP's) for specific topics in current research. The NRP's provide information on the objectives of a project and in some cases on progress and results. The NRP's from SSIE are more comprehensive since they encompass abstracts of all government funded research projects in the United States and some from other countries. The NRP's are not limited to USDA and the land grant university research projects. Projects identified in the CRIS abstracts also will be identified in the SSIE-NRP's.

Information packages available from SSIE that are pertinent to this project include

- Non-Point Water Pollution from Agricultural and Forest Land

- Waste Removal and Waste Treatment of Livestock

- Groundwater Contamination from Agricultural Sources

NRP's on these subjects were obtained and were used to provide information on current research.

Information on the SSIE informational packages can be obtained from:

Smithsonian Science Information Exchange
1730 M Street NW
Room 300
Washington, D.C. 20036

6.7 Canada

Individuals in Canada have been concerned with the problem of farm animal wastes for as long a period as those in the United States. Important contributions have been and continue to be made by Canadian investigators.

The Environmental Protection Service of Canada has published "An Annotated Bibliography of Farm Animal Wastes" (78) which is a very comprehensive summary of the existing literature throughout the world. Literature cited in the CRIS and WRSIC reports can be found in the Bibliography.

Reports on specific research projects are available periodically. Information on such reports and the Bibliography can be obtained from:

Environmental Protection Service
Department of the Environment
Ottawa KIA-OH3
Ontario, Canada

6.8 England

Considerable research and demonstration activities related to farm wastes have been conducted in England in the past decade. Results of their studies can be applicable to similar wastes in many of the Great Lakes states. Published reports of these studies have been cited in the WRSIC reports and the Canadian Bibliography.

Information on current projects and recent publications can be obtained from:

Farm Waste Unit
Agricultural Development and Advisory Service
Ministry of Agricultural, Fisheries, and Food
Coley Park, Reading, England

Section 7

ECONOMICS OF POLLUTION CONTROL

7.1 General

Adequate knowledge about the costs of animal waste pollution control related to either waste treatment and disposal practices or land runoff control practices is fragmentary. Both the actual costs to the producer and the ultimate costs to the consumer need to be known. Only recently have there been reasonable estimates of what the costs to the producers might be.

Most pollution control systems installed and operated by livestock producers are likely to increase the costs of livestock production. Economic tradeoffs may be possible if a pollution control device increases the efficiency or decreases the cost of one of the production components. In general, however, such tradeoffs are not likely to result in a net decrease in overall costs. A cost increase caused by incorporation of pollution control systems will reduce the net income of a livestock producer because there is no opportunity for an immediate offsetting livestock price increase.

Depending upon the magnitude of the cost increase, the livestock producer may chose to incur a reduction in net income or to discontinue business. Those who stay in business may increase their output to offset the increase in costs.

The influence of enviromental controls such as water pollution regulations can enhance economic growth if such controls are properly formed and adequately administered (79). Obsolete and improperly managed facilities may not be able to survive but new and better facilities will replace them.

Waste handling and disposal systems cannot be evaluated only by the level of investment or operating costs since they are an integral part of a total production system. However such items do permit a comparison of costs when pollution control systems are being considered. Unit costs are affected by the size of the overall operation with the largest per unit costs and therefore larger financial burden being on the small volume producer.

Available estimates of the costs required to achieve adequate water pollution control, especially to meet the EPA effluent guidelines are noted in the following paragraphs. It should be remembered that these estimates are based upon many assumptions regarding size, management, and feasible waste management approaches. They rarely represent data from actual livestock operations.

7.2 Nationwide

The total required investment to meet the proposed EPA effluent guidelines for the fed-beef, dairy, hog, lamb feeding, and poultry operations with surface water problems was estimated at $800 million (72) to control surface runoff from a 10 year 24 hour rainfall and $900 million to control the runoff from a 25 year 24 hour rainfall. The small livestock producers would have the highest per unit costs and probably would be affected the greatest by the need to install the runoff retention systems. For example, some 69 percent of the total estimated investment of $146 million in the fed-beef industry would be required of operators selling less than 100 head. In the hog industry, four fifths of the estimated investment of $280 million would fall on producers selling less than 500 head (72). For some operations, the cost of complying with the guidelines would be prohibitive. An estimated 4 percent of all dairy producers would have to move farms and lots to comply with the guidelines.

The decision by a livestock producer to invest in pollution control equipment and facilities, as well as the opportunity to obtain financing, is influenced by factors such as: equity position, future earning capacity, fixed investment in the livestock enterprise, and the importance of the livestock enterprise in relation to the total business (80). For most producers, the investments required to control surface water runoff generally are small in relation to the total investment in the farm operation. This holds for all producers, although the smaller producers will have a larger investment in pollution control equipment per animal than will the larger producers. It is unlikely that viable livestock operations would have difficulty in obtaining loans for pollution abatement systems (80). If a financing problem exists, it is likely to be for smaller producers who are in poor equity and low income earning positions.

Federal land banks, life insurance companies, and commercial banks are the major institutional lenders that make long-term loans for farm improvements secured by farm real estate. Commercial banks and production credit associations are important suppliers of short-term credit for farm investments. Farmers Home Administration (FHA) loans are available on both long-term and short-term bases to eligible farmers who are unable to get credit elsewhere on reasonable terms (81).

As with other farm improvements, installations of pollution abatement facilities would be an eligible purpose for FHA loans. Although pollution abatement is an eligible purpose for most lenders, many farmers may not be able to get financing for such facilities because they do not add to earning capacity. This is most serious for small or marginal farmers who may have all the debt they can handle.

Conditions that would make loans for pollution abatement more attractive to lenders include more technical assistance to the livestock producers, better estimates of costs and financial impacts on farm earnings, more cost sharing by the Government, and greater tax incentives (81).

There is no doubt that because of the increased costs associated with pollution control, the trend toward fewer but larger livestock operations will continue and possibly increase.

7.3 Poultry

About 92 percent of the egg production units and almost 97 percent of the volume are under confinement operations. A few laying hen operations have liquid manure handling systems that are not satisfactory. Point pollution problems are not regarded as serious. The small volume of production faced with serious point source pollution problems has led to the conclusion that neither egg supply or prices would be much affected by the EPA regulations (82).

Virtually all broilers are produced in confinement. Few point source pollution problems occur with these operations and there should not be much change in supply or prices as as a result of point source pollution restrictions (82).

About 89 percent of the turkeys are raised on the range. Many turkey growing ranges are faced with point source pollu-

tion problems. The incidence of potential problems varies widely between states. The costs of installing runoff control units to meet the EPA effluent guidelines was estimated to be about 30-40 cents per bird (82).

7.4 Hog

Most hog production comes from small enterprises on crop-livestock farms. Approximately 22 percent of the producers in the 15 major hog producing states were estimated to have a runoff problem requiring additional control. The investment cost per hog required to control surface runoff to meet the EPA guidelines was estimated to range from $61 per hog for very small producers to $4 per hog for extremely large producers (83).

About 60 percent of the producers needing runoff control are small volume producers with high unit costs. Many are marginal producers and an added high cost for pollution control would force many out of business. Thus regulations for controlling runoff would continue the change in structure of the hog production industry toward more larger volume producers.

Other estimates of the costs of hog feedlots to meet the EPA effluent guidelines indicate that the annual cost for effluent control would vary from about $3 per head for the smallest hog operation to about $1.23 per head for operations producing 900 hogs annually (84). Incremental costs for investments in runoff control lagoons and liquid dispensing equipment were estimated as varying from $2,050 for the operator producing 100 hogs annually to $5,100 for the operator producing 900 hogs annually.

Initial investments for feasible, available waste handling and management systems ranged from $1 to $12 per hog with an annual cost of $0.3 to $3 per hog for equipment and operation (81). A producer finishing 1500 pigs annually was assumed in estimating these costs. The waste systems considered included anaerobic lagoons, oxidation ditch, holding pits, and equipment to load and land spread the wastes.

7.5 Dairy

Milk is produced under a number of different situations throughout the United States. Climatic conditions determine

housing types and manure handling procedures. In the northern region, which includes the Great Lakes states, herds tend to be smaller and cows are housed in barns or closed shelters, particularly during winter months. Shelter facilities usually have outside lots associated with them. Manure is spread almost daily. About 38 percent of the farms in the northern region were indicated to have a runoff control problem.

In the northern region, it is estimated that the cost of controlling surface water runoff to meet the EPA guidelines would range from about $2800 to $3700 per farm with the unit costs ranging from $187 to $25 per cow (85). Costs expressed on a 100 pounds of milk basis were estimated as ranging from 13 to 93 cents per 100 kilograms (6 to 42 cents per 100 pounds). The largest unit costs occur on the smaller farms. As a result, the guidelines would hasten the trend toward larger, more efficient farms.

Incremental costs for effluent control for dairy farms were estimated to range from $1200 for a 25 cow herd to $6000 for a 1000 cow herd (84). Total investments per cow decreased as herd size increased and varied from $44 for 25 cow herds to $4.85 for 1000 cow herds. These costs were estimated to range between 1-4 percent of the total investment on the dairy farms.

7.6 Beef

In the major eastern beef producing states, which include the Great Lake states, there were about 179,000 beef producers in 1969. Feedlots with less than 100 head capacity produced 26 percent of the fed-beef marketed in these states. Between 25-50% of the lots in the eastern states have water pollution problems.

The highest investment per head of cattle to control runoff and meet the EPA guidelines are incurred by the small operations with land extensive open-lot systems located in the humid eastern beef feeding states. Estimates of new investments in runoff control systems ranged from $145 per head for lots with less than 100 head capacity, to $21 for lots with 100-199 head capacity, and to $3 per head for lots with over 1000 head capacity (86).

The incremental costs for beef cattle feedlots to meet the EPA effluent guidelines were estimated to vary from $2,050 for a feeder marketing 100 head annually to $63,500

for feeders marketing 40,000 head. The estimated annual costs ranged from $3.04 for the smallest feeder to $0.65 for the largest feeders (84).

A detailed analysis of alternative beef waste management systems is available (87). Cost and design information were obtained from feedlot visits and the literature. For large feedlots (20,000 head), the operating cost of a waste management system for an unpaved lot with pollution control was $0.133 per animal/day. The pen facilities were about 65% of the total investment cost, the runoff control system about 10%, and the solids handling about 25%. Thus the runoff control system cost was only a small fraction of the total waste management costs. Seven different waste management systems were evaluated and investment and operating costs estimated.

7.7 Current Research

Based upon Notices of Research Projects available from the Smithsonian Science Information Exchange, the following pertinent current research projects were identified:

- Smith, R.C., "Economic Analysis of the Impact of Pollution Control on the Poultry Industry in the Northeast", University of Delaware, Department of Agriculture and Food Economics. The economics of alternative techniques for disposing of wastes ranged from $16 to $29 per ton of waste material. Alternatives included disposal on the land, disposal to a landfill, and delivery to swine producers.

- Johnson, D.B., Connor, L.J., and Hoglund, "Economics of Livestock Waste Management in United States Fed Beef and Dairy Production Regions", Economic Research Service, Farm Production Economics Division, U.S.D.A., Michigan State University. A major objective is to evaluate the effect on production costs and investments for firms complying with animal waste regulations - progress not reported.

- Baker, M., "Economic Impacts of Water Pollution Control Programs on Cattle Feedlots", Nebraska Agricultural Experiment Station, Lincoln, Nebraska. A major objective is to assess the economic effects of water pollution from feedlots, to evaluate alternative means of pollution abatement and to estimate the potential economic impacts on the feedlot enterprise. The estimated annual cost for feedlot pollution control in the Papio watershed was estimated to be

$1.28 per head capacity with a range of $2.38 per head capacity for feedlots with under 125 head capacity to $1.12 per head capacity for lots with more than 2100 head capacity. Pollution control structures could require $13.23 investment per head capacity for feedlots with approximately 100 head capacity.

Section 8

AVAILABLE TECHNOLOGY

8.1 General

Solutions for satisfactory animal waste treatment and disposal will not be direct extensions of those that have been successful for industries and municipalities. Compared to municipal and industrial wastes, animal wastes are low volume, high strength wastes. When diluted, their characteristics may be similar to the concentrated solids from a municiapl waste treatment plant. However, because there are many alternatives to a water carriage system for the handling of animal wastes, there is no need to dilute the wastes to a level comparable to the concentration of municipal wastes. In addition to the difference in waste characteristics, available pollution control technology should recognize that inadequate labor exists at most livestock operations for complex treatment and disposal methods, that the wastes are generated at remote, diverse sources, that transporting the wastes to a central location for treatment and disposal is unlikely, and that no subsidy in the form of construction or operations grants is available to livestock operations for waste treatment and disposal. As a result, simple processes that are able to be integrated with existing livestock production methods and that utilize available equipment and personnel will be those that will be accepted and successful.

In spite of the fact that research on suitable animal waste management methods is of recent vintage, there are a number of processes that can minimize flagrant pollution problems and that can be integrated with modern animal production operations. No one treatment process or system will be the solution for all operations. A suitable system for a specific facility will be a function of the size of the operation, available land, topography and meterology, soil type, nearness to neighbors, and ability to maintain applicable product quality requirements. As with other industries, animal production operations should be located where minimum cost treatment facilities are possible.

Site selection requires attention. Until recently, many animal production facilities have been developed with little planning and concern for the nuisance and pollutional characteristics inherent with their operation. Many of the obvious pollution problems could have been prevented if the animal production facilities had been located in areas less susceptible to runoff, to uncontrolled release of wastes to streams,

and to air movements that carry odors to residential areas.

In the past decade, a variety of alternative waste treatment-handling-disposal systems have been identified and demonstrated. The intent of this report is not to provide a literature review of all possible processes but to indicate the importance of the processes in reducing water pollution and to indicate the effectiveness of available technology to cope with pollution problems. For the purpose of the report, the available technology will be grouped and discussed in the following manner:

a) technology that will affect the characteristics of the wastes so that the pollution potential of the wastes is decreased when they are disposed of on the land

b) technology that relates to the disposal of wastes on the land and its effectiveness in controlling water pollution.

The assumption that livestock wastes will be disposed of on land is implicit in this discussion. This assumption appears reasonable in light of the effluent guidelines proposed by the Environmental Protection Agency and because land disposal, with or without intermediate treatment, is a feasible method for the disposal of these wastes.

8.2 Waste Treatment Technology

As indicated above, this section will attempt to identify the technology that is available to treat livestock wastes prior to land disposal and which may affect the potential pollution of the wastes when disposed of on the land. There are many Conference Proceedings, technical reports, and summaries of available technology. The most comprehensive and concise is the Development Document prepared to identify the waste characteristics of the feedlot industry and applicable technology (77). This document contains information on twenty-nine separate processes with comments on the development status of a process and its reliability and applicability. This document should be consulted for details of specific processes. Tables 17 and 18 are from that document and identify the processes that were evaluated, their function and status, and the energy usage and by-products.

Many of the processes have not been used widely with animal wastes. All have had their potential evaluated

TABLE 17
END-OF-PROCESS TECHNOLOGY CLASSIFICATION (77)

TECHNOLOGY	APPLICATION			FUNCTION		STATUS			TYPE OF PROCESS	
				Treatment						
	Manure	Run-off	Con-tain-ment	Com-plete	Par-tial	BPCTCA	BATEA	Ex-peri-mental	Bio-chem-ical	Phys-ical Chem ical
Land Utilization	x	x		x		x			x	
Compost and Sell	x			x		x			x	
Dehydration	x			x		x	x			x
Conversion to Industrial Products	x			x				x		x
Aerobic SCP Production	x			x				x	x	
Aerobic Yeast Production	x			x				x	x	
Anaerobic SCP Production	x			x				x	x	
Feed Recycle	x			x				x		x
Oxidation Ditch	x				x	x	x		x	
Activated Sludge	x				x		x		x	
Wastelage	x				x		x		x	
Anaerobic Fuel Gas	x				x			x	x	
Fly Larvae Production	x				x			x	x	
Biochemical Recycle	x				x			x	x	
Conversion to Oil	x				x			x		x
Gasification	x				x			x		x
Pyrolysis	x				x			x		x
Incineration	x				x			x		x
Hydrolysis	x				x			x		x
Chemical Extraction	x				x			x		x
Runoff Control		x	x			x				
BLWRS		x		x				x	x	
Lagoons for Treatment		x	x		x	x			x	
Evaporation		x	x		x	x				x
Trickling Filters		x			x			x	x	
Spray Runoff		x			x			x	x	
Rotating Biological Contactor		x			x			x	x	
Water Hyacinths		x			x			x	x	
Algae		x			x			x	x	

TABLE 18
ENERGY AND NON-WATER QUALITY ASPECT
OF LIVESTOCK WASTE TECHNOLOGY (77)

Technology	Net Energy Usage	By-Product
Land Utilization	Low	None
Composting	Low	None
Dehydration	High	None*
Conversion to Industrial Products	Low	None*
Aerobic SCP Production	High	None*
Aerobic Yeast Production	High	None*
Anaerobic SCP Production	Low	None*
Feed Recycle Process	Low	Fiber
Oxidation Ditch	High	Sludge, liquid
Activated Sludge	High	Sludge, liquid
Wastelage	Low	None*
Anaerobic Fuel Gas	Low	Sludge
Fly Larvae Production	Low	Compost
Biochemical Recycle	Low	Fiber
Conversion to Oil	Low	Ash
Gasification	Low	Ash
Pyrolysis	Low	Ash
Incineration	Low	Ash
Hydrolysis	Low	None*
Chemical Extraction	Low	Liquid
Runoff Control	Low	Liquid, solids
BLWRS	Low	None
Lagoons for Treatment	Low	Sludge, liquid
Evaporation	Low	Sludge
Trickling Filters	Low	Sludge, liquid
Spray Runoff	Low	Grass, liquid
Rotating Biological Contactor	Low	Sludge, liquid
Water Hyancinths	Low	None*
Algae	Low	None*

*Note: Unless otherwise specifically indicated ash, salts or similar system residuals, if any, are not fully established at full scale.

through laboratory and frequently demonstration scale research. The available pollution control technology includes: land disposal, composting, dehydration, oxidation ditch, activated sludge modifications, wastelage production and refeeding, runoff control, lagoons for treatment, and evaporation. It is unlikely that evaporation is useful in the Great Lakes region to control pollution from livestock wastes. The other processes can be applicable to operations in this region and will be discussed briefly.

8.2.1 Liquid Processes

8.2.1.1 Runoff Control

Where animals are enclosed in the open, such as in a feedlot, runoff from these areas has the potential for serious waste pollution problems. The characteristics of the runoff and its volume will be related to the rainfall-runoff relationships that exist in a specific area. Proper site selection and the location of the feedlot can reduce the potential of water pollution significantly. A site should be chosen that is away from a stream or waterway. Animals in confined feedlots should not have free access to a stream since this will permit a natural stream to run through the lot and be the receptor of feedlot runoff. Drainage of feedlots through adjacent property, road ditches, or directly into natural streams or lakes is not desirable. Land area should be available so that drainage from the lots can be retained on property owned by the feedlot.

A critical feedlot management item is to divert all water outside the feedlot from flowing through the lot. All water from adjacent land and from roof drains should be diverted to avoid contact with the feedlot wastes. Diversion of extraneous water provides for a drier environment for the animals, possibly reduced odor, and a marked decrease in the volume of feedlot runoff to be managed. Even with proper site selection and lot management some runoff will result and will require suitable management.

The use of conventional waste treatment techniques for feedlot runoff and wastes will not be necessary because land is generally available for runoff control and waste disposal. The most practical approach for feedlot runoff

control appears to be a combination of protective dikes and levees to prevent the entry of rainfall from outside the feedlot area, plus collecting dikes, levees, and holding ponds to collect the rainwater falling directly on the lot (Figure 5). After collection and retention, some acceptable means of disposing of the liquid and solid material such as fertilization and irrigation on available pasture and cropland should be employed. Overflows from the retention ponds may be allowed only during times of rainfall that exceed the design criteria. Acceptable criteria appear in the effluent guidelines proposed by EPA.

The use of runoff retention ponds will change the characteristics of the livestock wastes. The volume of contaminated material will be greater by the amount of runoff water, there will be a greater amount of soluble contaminants, the solids can undergo anaerobic decomposition on the bottom of the ponds, and the size of particulate material can be reduced by microbial action. Perhaps the changes having the most significant impact on subsequent land disposal considerations are the large volume of liquid to be handled and the increased concentration of soluble contaminants.

The amount of land needed for disposal of feedlot runoff and wastes collected in a retention pond will be related to the possible accumulation of nitrogen and salts in the soil profile. In the areas with a low amount of rainfall and high evaporation rates, higher concentrations of these contaminants are in the runoff. Feedlot runoff in Texas had too high a salt content to be used as crop irrigation water (88). The electrical conductivity of the runoff was 6 to 10 mmhos/cm. Desirable water for irrigation did not have a conductivity exceeding 2 mmhos/cm. Because of the higher rainfall in the Great Lakes states, such a salinity hazard should be negligible.

Retention ponds will minimize the pollution caused by runoff from open animal production units. They should not, however, be considered as a sole source of treatment since their effluent does not meet requirements for discharge to receiving streams. With proper operation, runoff retention ponds are a very feasible waste management technique for confined, unenclosed feedlots. Their value has been proved in many cattle feeding states.

Grass terraces offer another opportunity for runoff control and can be incorporated with runoff retention ponds. Runoff flowing over grass or other land prior to retention ponds can be expected to be reduced in volume by soil per-

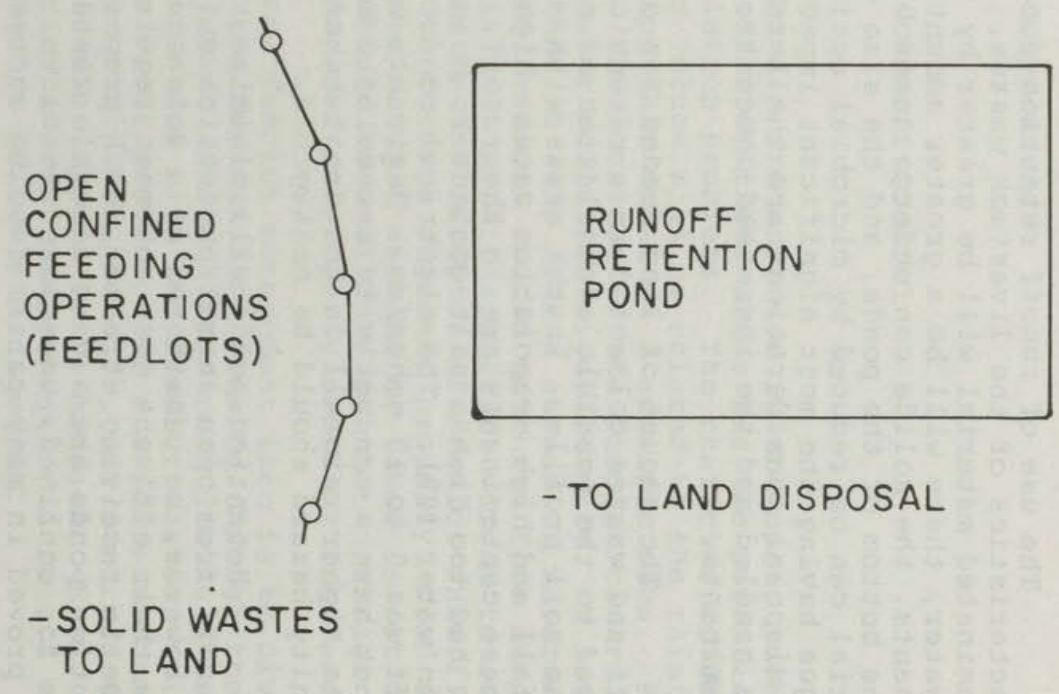


FIGURE 5
SCHEMATIC OF FEEDLOT RUNOFF
CONTROL SYSTEM

colation and in pollutional characteristics by solids removal on the land and by oxidation of some of the organic matter in the stubble or grass.

Runoff control systems using a settling channel, porous dams, and a detention pond for holding the liquid portion of the runoff have been investigated (89). The objective of these systems was to remove settleable solids from the runoff. Disposal of the settled solids was on adjacent fields using suitable equipment and the liquid in the retention pond was pumped onto adjacent cropland. The settling channels were successful with the heavier particles such as undigested grain settling out rapidly. The porous dams consisted of crushed rock supported by wood planking. Reducing the velocity of runoff flow was a low maintenance method of controlling the settleable solids content of runoff reaching a runoff detention pond. In a continuous flow system, up to 50% of the total solids in feedlot runoff were removed by a series of three porous dams. This approach is feasible for separating discrete particulate material from animal waste runoff. It may be less successful with wastes that contain fine particulate matter such as dairy or poultry manure slurries. Such fine matter may be difficult to remove from the settling areas and may plug the porous dams.

8.2.1.2 Enclosed Operations

Many livestock operations utilize enclosed confined facilities in part or all of their operations. Enclosed operations are not subject to the probabilistic nature of precipitation events and a greater variety of treatment technology is possible. The ultimate disposal of wastes generated at these facilities is on the land. Closed system recycle opportunities also are increased because of better control of the wastes.

A waste management system for enclosed confined facilities will deal with wastes having a consistency of "as defecated", as a liquid, or as a "dry" material. The most feasible approach will depend upon the nature of the waste, and labor requirements and availability. Justification of a liquid manure system frequently is done on the basis of reduced labor demand. To obtain the benefits of improved labor distribution, reasonably long storage periods and a planned disposal schedule must be available and maintained. The long storage can be under both anaerobic and aerobic conditions. Schematics of possible

waste treatment and disposal alternatives are noted in Figure 6.

Daily manure cleaning and spreading on the land or daily manure cleaning, manure storage, and intermittent disposal on the land are feasible manure handling procedures at many livestock operations. The simplest form of waste handling can consist of a tractor with a rear or front mounted blade or bucket and a ramp for pushing the manure into a spreader. This system requires daily spreading. Difficulties in disposing of the wastes on the land result when the land is wet or frozen or when crops are being grown on the land. Labor is distributed throughout the year. A holding tank permits flexibility during periods when the wastes cannot be disposed of on the land. In northern climates, holding capacity of from three to five months is suggested. These holding tanks are not aerated and odorous gases and compounds will result due to the anaerobic conditions in the tanks.

Livestock waste storage in holding tanks is not a treatment process. Because of the biodegradable nature of the wastes and lack of aeration in the tanks, anaerobic degradation of the material will occur. Such storage is not a controlled anaerobic digestion system but storage under anaerobic conditions that cause partial decomposition and odor. The odor nuisance is caused more by the movement of the stored slurry, i.e., pumping and cleaning, than by the fact of its storage.

The changes in waste characteristics that occur in anaerobic storage include an increase in soluble contaminants and a possible increase in the volume of the material requiring disposal. The latter occurs because of the necessity of adding water to the accumulated wastes to facilitate their removal by pumping. Generally the solids content must be in the range of 10-15% to accomodate removal by pumping. Where solids removal can be accomplished by solids removal equipment such as front end loaders, the additional water is not necessary and the waste volume need not be increased.

In northern climates, it may not be possible to spread dairy manure every day and manure stackers can be used to stack the dairy manure in a walled storage area during the winter months. This results in an exposed holding area with both solids and drainage to take care of. A runoff retention pond should be used with such stacking systems to assure that any drainage is kept from nearby

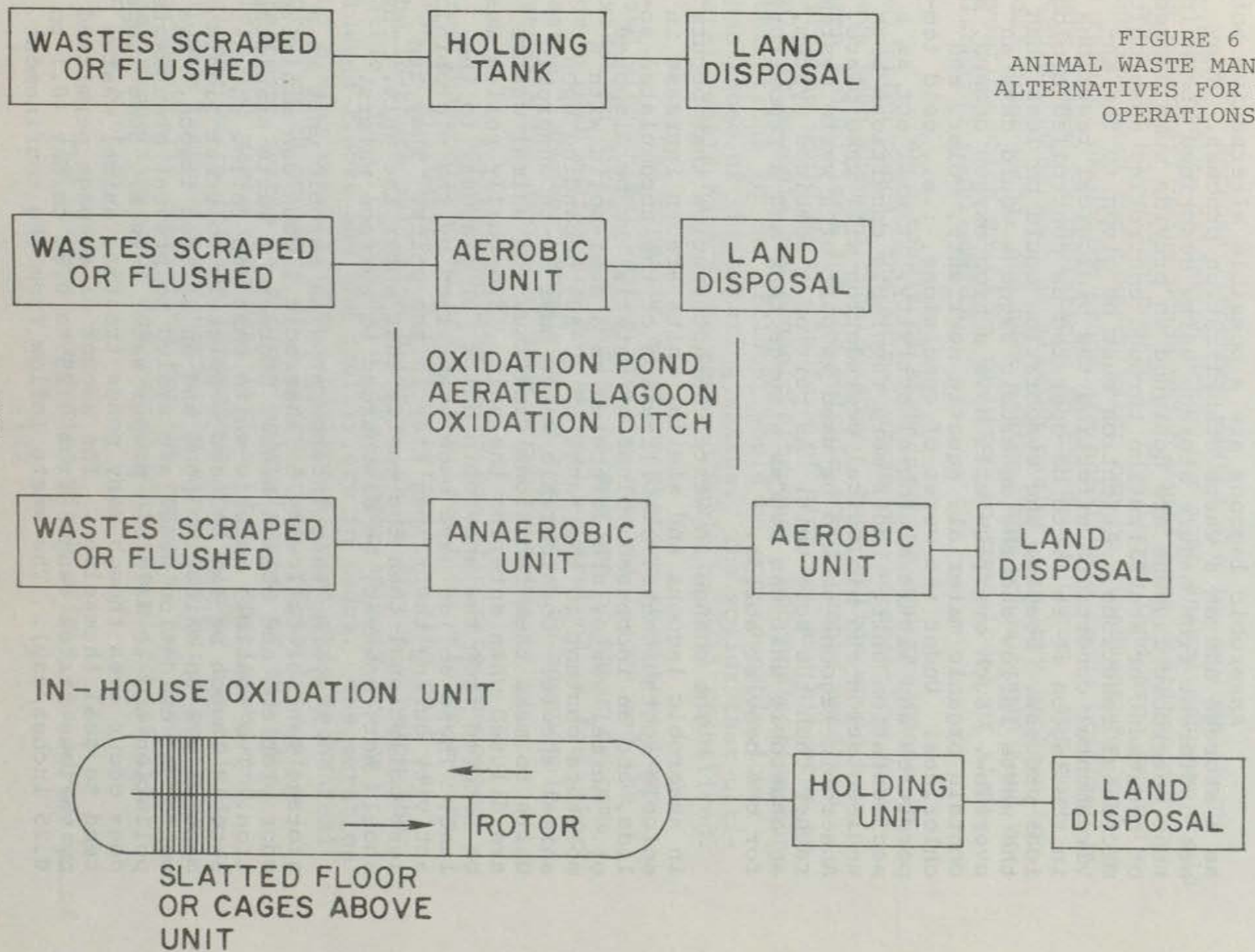


FIGURE 6
ANIMAL WASTE MANAGEMENT
ALTERNATIVES FOR ENCLOSED
OPERATIONS

streams.

Anaerobic lagoons are a possible alternative both as a storage and as a waste stabilization process. They are different from manure storage units described above in that anaerobic lagoons are designed to provide some degree of treatment or stabilization through controlled anaerobic microbial reactions. Attention must be given to the environmental conditions affecting the biological reactions in the lagoon if it is to be operated as controlled biological process. Seepage and evaporation could be less than the waste inflow and the anaerobic lagoons would have an overflow. Such overflow will have a high oxygen demand, and contain organic material, aquatic nutrients, color, and chlorides. Under the worst of conditions, i.e., cold temperatures and little microbial activity, it will act as a sedimentation unit. Under more favorable conditions both sedimentation and biological degradation will take place. Anaerobic lagoons should be used as part of a waste system rather than the sole unit. As part of a combined system, an anaerobic unit can act as a surge tank and a repository for the heavier solids.

The change in waste characteristics that occurs in anaerobic lagoons and similar units are an increase in soluble contaminants, stabilization of the accumulated solids, and an increased volume of material to be disposed of. Animal wastes are defecated as a semi-solid with a moisture content in the range of 70-80% depending upon animal specie. Considerable water must be in anaerobic units to make them functional and thus the animal wastes are diluted when entering the unit. Eventually solids must be removed from the anaerobic units for disposal to the land. These solids have been stabilized by the microbial activity during their long stay in the units. They are less pollutional than the wastes that enter the unit. They should not, however, be discharged to surface waters.

A system that incorporates an aerobic unit as a storage tank or following an anaerobic lagoon can minimize odor problems and obtain higher degrees of waste stabilization. Two feasible aerobic units that can follow an anaerobic lagoon or can be used separately to treat liquid manures are an oxidation pond and an aerated lagoon. The bottom of oxidation ponds are sealed to prevent groundwater pollution and to maintain proper water levels. Some seepage does occur even though many ponds treating animal wastes tend to seal themselves. The amount of seepage permitted by various states ranges from 0.25 to 0.63 cm/day (0.1 to 0.25 inches/day). The waste inflow from the confinement

operations may not be adequate to overcome permitted seepage losses. An additional source of water would be needed to maintain proper water depth.

Because livestock wastes are low volume, high solids waste, oxidation ponds for most animal manures could be non-overflowing ponds which would have distinct advantages for pollution control. The large land area needed and the need for make-up water can be disadvantages and as a result, oxidation ponds are not likely to be utilized by many livestock operations in the Great Lakes. For more liquid wastes such as milking parlor, poultry processing, milk processing, and duck wastes, the ponds would have an overflow which would be required to meet local or state discharge requirements.

As a result of stabilization and treatment in oxidation ponds, the animal waste characteristics would have been changed considerably. Many solids would have been solubilized and the soluble constituents oxidized. Nitrogen would be in the form of nitrites and nitrates as well as ammonia and organic nitrogen. BOD and solids concentrations in the effluent would be considerably lower than those of the entering wastes. The solids that ultimately would have to be removed would be well stabilized.

The types of aeration systems that have been used with animal wastes are the aerated lagoon and the oxidation ditch. The design parameters of these systems for animal wastes are not as well understood as for other wastes. However, knowledge is available to adequately estimate the design of such aerobic units.

Aerobic holding units can eliminate the odors from anaerobic units and at the same time oxidize a portion of the wastes. A common aerobic holding unit is the in-house oxidation ditch which is directly below the cattle, hogs, or poultry. The larger animals are on slatted floors and the poultry are in cages. The aeration and mixing results from a horizontally mounted mechanical rotor which operated at a fixed speed. This type of a treatment system is a close approximation of the ideal microbial waste treatment system, i.e., continuous and uniform waste loading throughout the unit by the animals, minimum temperature fluctuation since the building is environmentally controlled for the animals, and good and continuous mixing of the wastes with the microbial life in the system.

The in-house oxidation ditch offers the advantage of

inexpensive construction since it is part of the confinement building, minimum odor problems since it is designed to be aerobic, and fewer handling problems since the wastes are pushed through the slatted floor to the ditch. A large number of livestock oxidation ditches are in operation in the United States. The majority are for swine operations although some are being utilized for beef cattle and poultry operations.

When land disposal of livestock waste occurs, constraints on the amount that can be applied to an unit of land will be related to either the quantity of liquid that can be absorbed on a routine basis or on the quantity of nitrogen that can be added without resulting in excess nitrogen in the runoff and soil water percolate. Aerobic liquid waste treatment units such as the oxidation ditch offer opportunities to control the nitrogen content of waste by designing and operating the units to achieve controlled nitrification and denitrification. Nitrogen losses have been observed to occur in oxidation ditches treating animal wastes. From 50 to 80% of the total Kjeldahl nitrogen loading to a ditch treating swine wastes was lost during treatment (90). Other investigations have observed similar results. Between 25-30% of the input total nitrogen is normally lost during the oxidation ditch treatment of livestock wastes. Operational procedures to achieve desired nitrogen losses between 25 and 80% in an oxidation ditch are under study (91). Nitrification-denitrification has been identified as the most feasible nitrogen control process that can be applied to livestock wastes (92).

Experience with aerobic treatment units for animal wastes has shown that these processes can have high BOD removal efficiencies. The BOD in the effluent from the units is almost entirely due to the oxygen demand of the solids in the effluent and to nitrification. The resultant effluent will still have a significant residual total BOD and is more suitable for land disposal than for disposal to a receiving stream.

One of the more interesting challenges is to design these aerobic systems to meet a specific effluent standard which is governed by land disposal requirements rather than by stream criteria. Because land disposal is an integral part of animal waste systems, conventional solids removal systems need not follow these aerobic systems.

8.2.2 Solid Waste Processes

Animal wastes are defecated in a semi-solid form and there is considerable logic in treating and disposing of these wastes as a semi-solid or solid material rather than adding water for handling and treatment or disposal. The addition of water requires the treatment of both the animal wastes and the contaminated water. When considering animal solid waste disposal, it is important to obtain the initial wastes in as dry a condition as possible to facilitate most of the possible solid waste disposal methods. This may mean separation of liquid and solid wastes at the source or having natural drying conditions occur.

Drying and composting are feasible with animal wastes. Poultry waste, because of its lower initial moisture content, requires less energy to dry than will wastes containing a higher water content. In-house drying of poultry wastes by greater air circulation, by the addition of heat, or by both offers the possibility of least cost drying systems for these wastes. The success of drying and composting requires a market or ultimate disposal point for the product.

A significant volume reduction will take place when livestock wastes are dried or composted. The moisture content can be decreased to the 40-50% level during composting and to as low as 20% during drying depending upon the amount of heat used and the objective of the drying process. Volatile compounds would be lost in both processes. Microbial reactions in composting will decrease the carbon content as the wastes are degraded. The nitrogen content would be decreased as the ammonia is lost by aeration and the high temperatures. The resultant product of both processes is smaller in volume and easier to handle than the initial livestock wastes.

Composting offers an opportunity to recover and reuse a portion of the nutrients as well as the organic fraction in animal wastes. Such wastes can be composted alone but also can be combined with wastes that may be nutritionally unbalanced, i.e., a high carbon-to-nitrogen ratio, such as sawdust, corn cobs, paper, and municipal refuse.

The fundamentals of composting involve intimate mixing of the wastes, small particle size, adequate oxygen for the microbial degradation of the wastes, adequate time to accomplish the composting, and adequate moisture. The composting can be done in open windrows or in enclosed environmentally controlled units. With the use of the controlled

units, composting can be accomplished in 5 to 7 days, while in open windrows, it may take 1 or 2 months to produce satisfactory compost. The major objectives in composting are to stabilize putrescible organic matter, to conserve as much crop nutrients and organic matter as possible, and to produce a uniform, relatively dry product suitable for use as a soil conditioner and garden supplement.

Composting has been technically successful with poultry, beef, and dairy cattle wastes. Economically, the process has not been as successful. A suitable market must be available before composting can be attractive as a method for the disposal of agricultural solid wastes. Without a suitable market most of the original dry matter remains for further disposal. Composting can be feasible for specific animal production units and in unique regional situations. As yet, it is not a process that can adequately dispose of the volume of animal wastes generated at livestock operations.

Composting can be considered as a possible process to be used prior to land disposal of livestock wastes. However, the costs of the operation as well as the technical and personnel needs do not make composting attractive as a pretreatment process prior to land disposal.

The fundamentals of drying or dehydration involve the addition of heat and/or air and mixing of the waste. Drying of poultry manure directly in the poultry house by increasing ambient air movement has been investigated and found to be a feasible process prior either to land disposal or to further drying in commercial units for re-feeding or other end-product uses. Commercial driers are available and have been used with a number of animal wastes. The gaseous effluent from the high temperature drying can be a source of odors and nuisance complaints. Satisfactory after-burners or other air pollution control equipment may be necessary.

Drying and dehydration are not disposal processes in themselves but are used to produce an end product suitable for satisfactory disposal either as a soil conditioner or for re-feeding to reuse a portion of the remaining nutrient value. The marketing potential of dehydrated animal wastes is unknown. Without a suitable market, high temperature drying is unlikely to be competitive with other treatment and disposal methods.

Incineration offers the possibility of disposing of a considerable fraction of animal wastes by combustion. It is not a total disposal method since about 10 to 30% of the

initial waste weight will remain as ash. Incineration is one of the more expensive processes that can be used with animal wastes, especially since the incinerators normally require air pollution control units to handle the gaseous and particulate material in the exhaust gases. As a result, incineration has not been applied to livestock wastes on any large scale.

8.2.3 Recycle and Reuse

A hydraulic manure-transport system utilizing recirculated treated effluent has been developed for swine wastes (93) and has been evaluated at a 700 head operation. The manure is flushed hourly to an anaerobic lagoon. The lagoon effluent is pumped back to the swine building and reused for flushing the gutters. The lagoon liquefies the manure and eliminates hauling and land application of solid manure. Sprinkler irrigation was used to control the lagoon level and dispose of excess liquid. The lagoon volume had sufficient volume to store excess liquid during periods when land disposal was not feasible. The result is a closed recycle system with minimum labor and no water pollution. Approximately three acres of cropland were required for the disposal by irrigation.

Livestock wastes contain metabolizable energy and nutrients. A number of studies have shown that wastes from livestock can be incorporated into feed rations for other animals without adversely affecting animal growth or health. The use of wastelage, cattle manure which has been removed daily, blended with hay, and stored as silage before use, has shown promise as a portion of animal feed (94). Dried poultry wastes successfully can be used as part of the feed rations for poultry (95, 96) with optimum values being about 10-15% of the ration. Experiments feeding the mixed liquor from an oxidation ditch treating swine wastes to swine have indicated the possibility of such an approach (97). The oxidation ditch mixed liquor contributed vitamins, amino acids, and minerals to a swine diet and provided the water in swine diet. Under the conditions of the experiments, no oxidation ditch effluent needed to be discharged thus approximating a closed cycle waste handling and disposal system.

The possibilities of the reuse of animal wastes as an animal feed supplement have been discussed (98). When nutritional principles are followed, a portion of animal wastes can be used as a feed supplement for animals. Ques-

tions related to the transmittal of drugs, feed additives, and pesticides to the second animal and to the agricultural product remain to be clarified.

The reuse of animal wastes in animal feed rations is not an ultimate disposal method since a portion of the waste is not able to be utilized by the animal and will remain for disposal. Recycling of wastes in this manner cannot go on indefinitely. There always will be wastes to be treated and disposed of from animals being refed processed animal wastes as part of their food. A completely closed loop cannot be envisioned with such an animal waste recycle approach.

Many other methods have been suggested for the conversion of livestock wastes into useful products such as protein for animal feed, chemicals, building material, and fuels. None have had nor are likely to have a significant impact on the livestock industry. A significant reduction of the pollution potential of livestock wastes by reuse or recycling methods will require the development of either a high volume, low cost use or product or the discovery of some presently unknown economic value in the wastes sufficient to pay for processing large volumes of the wastes. The most feasible reuse and recycling method for livestock wastes remains disposal on the land and incorporation of the waste nutrients in a crop usable for animal or human food.

8.3 Land Disposal Technology

Land is a non-point source of water pollutants primarily by erosion. The application of livestock wastes to land can increase the potential pollution if good land management practices are not followed. When these wastes are properly applied, land disposal of livestock wastes can be an acceptable practice.

The land represents an appropriate disposal site for livestock wastes as well as an opportunity to manage wastes with a minimum of adverse environmental effects. Manure has been applied to land for both disposal and its fertilizer value for centuries. However, the land cannot be utilized as a neglected waste sink. The soil and the wastes must be managed as a total system to avoid undesirable environmental problems.

Each soil will have a maximum capacity to assimilate wastes and renovate wastewaters. The soil waste assimilative capacity is related to the characteristics of the soil, crops to be grown, environmental conditions, characteristics of the wastes, and to the microbial, chemical, and physical reactions that take place in the soil. Dimensionally, the units of the soil waste assimilative capacity should be the quantity of a critical waste parameter applied per unit volume of soil infiltrated. Examples could be gallons of wastewater or pounds of nitrogen applied per cubic foot or per foot of soil depth per unit time. At a minimum, the soil waste assimilative capacity and hence the soil loading rate should be stated in terms of the quantity of waste applied per unit time per unit surface area.

Current waste application rates are in terms of tons of wet manure or inches of liquid per unit time. While these rates may provide information on operating disposal rates, they provide little understanding of the fundamental, controlling parameters. In addition, investigations on the land disposal of wastes rarely are comprehensive enough to permit adequate correlation between land disposal rates and subsequent runoff and groundwater pollution problems.

Some information on land application rates of manures has been presented in Section 4 - Surface and Groundwater Contamination. Additional information on land disposal of livestock wastes, site location of livestock operations and waste disposal, and on suggested guidelines is presented in this Section. No attempt will be made to discuss the equipment associated with the land disposal of wastes.

8.3.1 Manure Disposal

The distribution of livestock waste on land is the most common disposal method for these wastes. They may be applied to the land as a source of nutrients and organic matter or as maximum feasible loadings for disposal under conditions that do not create a pollution problem.

The methods of applying livestock wastes to land are by surface application or by soil injection of slurries. Surface application can be followed by immediate incorporation into the soil, no incorporation, or incorporation at a later date. The frequency of spreading livestock wastes on land varies, the range being from daily for many dairy operations to intermittently for poultry and beef cattle

operations. The preferred method for reducing the pollution potential from such disposal is to incorporate the wastes with the soil as soon as possible after disposal. Wastes applied by irrigation cannot be incorporated into the soil immediately because of wet conditions. The surface application methods include tank spreaders and various liquid irrigation systems. The soil injection systems are the best pollution control systems since the waste is immediately incorporated in the soil, virtually eliminating the possibility of polluted runoff.

The maximum or best rates of livestock manure disposal on land have not yet been established. A single nationwide or statewide recommended rate is not reasonable because the best rates will be location specific, depending upon soil type and properties which can vary widely. The livestock wastes should be added to a soil at a rate that the growing crop can utilize the nutrients in the wastes. The suitable rate of manure application is determined by the ability of the soil-crop combination to immobolize and utilize the nutrients in the manure. The greater the crop requirements for the plant nutrients, the greater the amount of manure that can be applied. Nitrogen is the nutrient of greatest concern from the potential pollution standpoint since it is soluble as nitrate and readily moved in runoff and leachate. Most agricultural soils have a large capacity to hold phosphorus. If large additions of nitrogen are not balanced by incorporation into growing crops, by denitrification losses, or by ammonia volatilization losses, the soluble nitrogen concentration in leachate and runoff will increase.

Many of the studies investigating the land disposal of livestock wastes have evaluated the effect of manures on crop yield and have identified disposal rates that did not result in crop inhibition or decreased yields. Information relating disposal rates to water pollution problems rarely are part of such studies and interpretation of the results in terms of potential or actual water pollution problems is difficult.

Intermittent waste applications preserve the effectiveness of the soil to assimilate the wastes. Too frequent or too heavy a waste application can decrease the rate of infiltration and the oxygen input to the soil. For best results, concentrated organic wastes such as manures should be injected directly into the soil as soon as possible by discing or plowing.

Poultry manure and broiler litter have been applied to

land under conditions that have caused soil pollution and related problems. Where manure-litter has been returned to the soil at rates exceeding 22 metric tons/hectare (10 tons/acre) annually, problems of excess salts and a chemical imbalance have occurred in the soil and problems of nitrate toxicity and grass tetany have occurred in the grass pasture (99). As a consequence of these potential health problems, broiler litter disposal rates greater than 9 metric tons/hectare/year (4 tons/acre/year) are not recommended on fescue pasture systems (100).

The plow-furrow-cover method was used to compare the disposal of poultry manure at rates of 0 to 100 metric tons per hectare (0 to 45 tons/acre) (101). The tests were on a loamy soil with a clay content of 15 to 33%. Soil and soil water analysis indicated that potassium, magnesium, and sodium were mobile and moved downward through the soil. The soil did not have a cover crop in these experiments. Ammonia nitrogen existed in the soil during the winter but generally was oxidized to nitrate nitrogen in the warmer months. Nitrate nitrogen in the soil was in low concentration shortly after manure application but increased to high concentrations in the manure application zone. Nitrate nitrogen concentrations as high as 700 mg/l were found in the soil water. With time and rainfall, the nitrate nitrogen was distributed throughout the soil profile and by the end of the experiment had leached below the lowest sampling point. Only very small amounts of phosphate were found in the soil water. Chloride and sulfate ions were found in the soil water at all depths. It was concluded that the disposal of poultry manure in these soils should be less than 34 metric tons/hectare (15 tons/acre) of dry material because of nutrient contamination of the soil water.

In Kansas, maximum yields of irrigated corn silage occurred at beef cattle manure application rates of between 225 and 290 metric tons/hectare (100 and 130 tons/acre)(40). Larger applications depressed yields due to accumulation of soluble salts in the soil. Potentially toxic accumulations of ammonium ion were found in the surface 30 centimeters (12 inches) of soil. Continued accumulations of large amounts of salts appeared to threaten infiltration and water movement in the soil profile. The irrigation runoff from the waste disposal plots contained COD concentrations of 10-80 mg/l, nitrogen about 15 mg/l, and phosphorus about 1 mg/l (102). Stormwater runoff from these plots had COD concentrations of 150 to 400 mg/l, nitrogen concentrations of 10 to 40 mg/l and, phosphorus concentrations of 1 to 3 mg/l.

Anaerobic livestock lagoon effluent was disposed of on grass covered soil and resulted in COD, phosphorus, and nitrogen reductions of 95%, 99%, and 80% respectively during the summer in Iowa (103). The loading rates ranged from 33 to 76 centimeters (13 to 30 inches) of lagoon effluent. Insufficient data was presented in the article to develop application rates in terms of the amount of a potential contaminant per unit area of land. The reductions were determined based upon samples of the soil percolate at depths from 7.6 to 76 centimeters (3 to 30 inches).

The amount of nitrogen that can be utilized by a crop places a limit on the quantity of wastes that can be applied to a given soil. The quantity of wastes to be applied can be increased if the excess oxidized nitrogen that is produced were denitrified in the soil. An engineered soil system to remove the excess nitrogen, phosphorus, and assure the decomposition of organic matter has been developed (104). Known as the barriered landscape water renovation system (BLWRS), the system consists of a mound of soil underlain by a barrier impervious to water. The barrier extends beyond the mound under the level soil at the edges. A thin bed of limestone or slag may be placed on the top of the mound and the wastes are spread over the top of the mound. As the water percolates through the mound, the organic particles are filtered out and decompose. The filter bed and the soil remove the major portion of the phosphate. The soluble organics and inorganics move into the aerobic soil zone where the ammonium ions are held in the exchange complex of the soil until they are nitrified. Most of the soluble organic matter is oxidized in the aerobic soil. The downward movement of the nitrates is stopped by the barrier and forced to move laterally through the anaerobic soil perched on the barrier. In the anaerobic zone, denitrification occurs. Carbon sources can be added to the denitrification zone, if needed, in the form of inexpensive organic matter such as grains. The renovated water moves from the edges of the barrier into the adjacent soil.

This barriered landscape water renovation system has been used with swine wastes. The renovated water has been shown to contain less than 2 mg/l of nitrogen and 0.07 mg/l of phosphate which amounted to 99.5% nitrogen removal and 99.8% phosphate removal (104). The long term evaluation of this system is needed to assess its actual potential for livestock operations.

8.3.2 Liquid Waste Disposal

Liquid livestock wastes, such as from runoff retention basins, oxidation ditch mixed liquor, or diluted manure slurries can be applied to the land using liquid waste irrigation systems. Ridge and furrow irrigation, spray irrigation, and overland flow irrigation can be considered for these liquid wastes. With the ridge and furrow method, liquid wastes are discharged into furrows and into the soil. Spray irrigation is the controlled spraying of liquid on the land. Lightweight metal or plastic pipes are used to transfer the liquid wastes to the disposal field. Overland flow irrigation is the controlled discharge of the liquid on the land with the liquid flowing downslope over the land surface.

Highly dilute wastes are required for these systems. Agricultural wastes applied in this manner have been food processing plant wastes, milk plant wastes, and similar liquid wastes. No references could be found to indicate the use of these methods with treated or untreated livestock wastes. However, when good land management and waste disposal guidelines are practiced, these methods have a potential use with certain types of dilute livestock wastes.

Where liquid livestock wastes are disposed of by irrigation, care must be taken to avoid subsequent runoff from the disposal areas. Tail water retention ponds can be used to avoid such possibilities. Application rates which do not result in subsequent runoff are the preferred way to avoid surface water pollution.

Although the disposal of liquid, untreated livestock wastes by irrigation is a possibility, it may not be the most desirable disposal method in the humid Great Lakes states where there is adequate precipitation and where odor problems are of concern. Direct land application of livestock wastes and rapid incorporation into the soil is a more suitable way to dispose of these wastes on the land.

8.4 Management Considerations

Soil eroded from land used for the disposal of livestock wastes will carry a portion of the nutrients, organic matter, and bacteria that were in these wastes. Proper erosion control can reduce the amount of contaminants entering surface waters from the lands.

Soil erosion and contaminant yield is a function of rainfall, soil characteristics, land slope length, steepness, and cropping practices. While little can be done to change the amount, distribution and intensity of rainfall, measures can be taken to reduce the ability of rainfall to cause erosion. Examples include decreasing the amount and velocity of overland flow and decreasing the impact of the rain on the soil. Both flow and impact can be reduced by maintaining vegetative cover on the land.

The incorporation of crop residue and animal wastes in the soil increases soil porosity and aggregation and decreases runoff. Studies in the northeast (48) have indicated that continuous corn grown with mineral fertilizer but without manure additions resulted in increased soil loss and surface runoff. The incorporation of crop residues and manure in the soil increased water infiltration and reduced surface losses of water and erosional losses of nutrients.

Slowing runoff by vegetative cover and land modifications (terraces and strip crops) offers opportunities to provide greater infiltration time, conserve moisture, and decrease erosion. Practices to conserve soil and water also reduce the pollutants in agricultural runoff. These practices include:

- minimum tillage on slopes
- use of terracing, strip cropping, contouring, and diversions
- early growth of crops
- sod crops in rotation
- avoid bare land surfaces
- spread manure on growing crops or stubble rather than on bare fields
- avoid application of manures on slopes capable of rapid runoff
- apply livestock wastes on the land uniformly and in accordance with crop requirements

A combination of livestock waste treatment-management-disposal methods are available for confined livestock pro-

duction operations. Where livestock are produced on pasture or range, good management is the most appropriate approach to avoid water pollution due to runoff. In the February 14, 1974 regulations for the feedlot industry, a feedlot was defined as a livestock operation where crop or forage growth or production is not sustained. With dispersed livestock operations, such as those on pasture, the discussions to the regulations note that "the combined effect of soil and vegetative assimilation of manure and the lower rate of manure deposition per unit area could reasonably be expected to preclude any significant pollution problem". This statement infers that range or pasture livestock operations are not causes of significant water pollution. Data to confirm or refute this statement is scarce. Under all circumstances, including pasture operations, it is advisable to apply the best pollution control management options.

In range or pasture livestock production, manure is deposited directly on the land by grazing animals. Even though a relatively large land area may be available to pastured animals, they tend to concentrate around resting, watering, and feeding sites. Good drainage around these sites is desirable. Streams in the pasture or range land also offer an opportunity for direct deposition of livestock wastes in surface waters.

The good practice approaches which will minimize water pollution from pasture and range have been identified (5) as:

- maintain an adequate land-livestock ratio; avoid concentrations of animals that will create holding rather than grazing areas

- maintain a productive forage on the pasture land to retard runoff, entrap animal wastes, and utilize nutrients

- plan a stocking density and rotation system of grazing to present overgrazing and erosion

- locate feeders and waterers at a reasonable distance from streams; move them often enough to avoid creating erodable paths by the livestock

- provide a sufficient land absorption area downslope from feeding and watering sites, preferably with a filter strip of forage between such sites and streams

- provide limited access to streams and ponds; pump water to the animals where concentration of animals or characteristics of the land present potential pollution problems.

The pollution potential from wastes of livestock on pasture increases if the land topography favors a high rate of runoff or if grazing patterns are such that wastes accumulate in areas near surface waters. Limited data are available on the actual impact of livestock wastes on stream water quality in grassland or pasture. The water pollution potential from animals grazing in pastures and grasslands has been noted in Section 4 and should be minimal under good range management practices and close to "background" or "natural" levels.

Site selection is an important factor in controlling pollution problems from livestock operations. A site should be selected which will minimize water pollution and be adaptable to necessary controls. A succinct summary of the basic factors to consider in locating a beef cattle feedlot is available (105). Although this report was prepared for cattle feedlots, the basic concepts are applicable to any livestock operation. Discussion is provided on spatial requirements, topographic features, microclimates, soils and geologic structures, and social considerations. Specific guidelines for runoff control are included as are suggested land areas for liquid and solid waste disposal and ranges of application rates.

8.5 Current Research

The Notices of Research Projects (NRP's) available from the Smithsonian Science Information Exchange and the CRIS Abstracts from the U.S. Department of Agriculture were reviewed to ascertain the type and number of research projects dealing with livestock waste management technology. During the 1971-73 period, about 80 projects were in existance that dealt with some aspect of such technology. These projects were supported by OWRR-USDI, EPA, CSRS-USDA, ARS-USDA, various state governments, and the Canadian government.

Some of the projects were completed by the time of this report. Results based upon project completion reports and published articles from a few of those projects appear in this Section.

Over 60 percent of the projects dealt with some aspect of waste treatment, primarily liquid waste treatment by oxidation ditches, aerated and anaerobic lagoons, and comparable processes. No project had as its objective the treatment of livestock wastes for discharge to streams. Land disposal was considered as the disposal site in these projects. These projects investigated liquid waste treatment as a method to control odors, to reduce the pollution potential prior to land disposal, and to facilitate overall waste handling and management in the livestock operation.

Runoff control and management projects were only a small proportion of total number of projects, about 5%. Project areas with at least 10% of the projects included land disposal of livestock wastes, livestock wastes as a feed supplement, and utilization methods for the wastes other than land disposal. Only one project existed in the following areas: pyrolysis, methane production, and pathogen survival.

The majority of the research was being conducted at Universities and USDA field stations. Only a few of the NRP's and abstracts provided more than cursory information on the progress of the projects. No evaluation of the project results could be obtained.

Only one or two projects appeared to develop information that ultimately would relate waste management technology to land disposal and subsequent unit runoff rates. The projects appeared adequate to determine the technological feasibility of the processes being evaluated. Few projects indicated that adequate economic data also was being obtained so that the economic feasibility also could be assessed.

The projects appeared to be the result of individuals within a single discipline although a few had objectives that appeared to be comprehensive and had interdisciplinary possibilities.

Section 9

NEED FOR ADDITIONAL ACTIVITIES

9.1 Summary

The specific role of animal wastes and intensive animal feedlots as they affect the water quality of the Great Lakes remains unclear. On the land tributary to the Great Lakes, animal production exists as a part of a combined crop-livestock production system and land is available for the disposal of livestock wastes. Based upon available information, there are relatively few livestock operations that can be classified broadly as an intensive animal feedlot in terms used in the Environmental Protection Agency regulations, i.e., an area where livestock are fed "at the place of confinement and crop or forage growth or production is not sustained in the area of confinement".

In the Great Lakes Basin there are about 130,000 livestock operations with sales of at least $2500 per year and an even larger number if smaller operations were to be included. In comparison, there are approximately 190 beef cattle feedlots having a capacity of 1000 head or greater in all of the states adjacent to the U.S. portion of the Basin. There is no readily available data on the numbers of the livestock operations that fall within those required to obtain NPDES permits, i.e., 700 dairy cattle, 2500 swine, 10,000 sheep, 55,000 turkeys, 5,000 ducks, or poultry operations using liquid systems; however the numbers of such large operations in the Basin should be extremely small. Thus, water quality problems in the Basin due to intensive animal feedlots do not appear significant.

Where such intensive animal feedlots are located, the potential for surface water pollution due to runoff can exist. Technology is available to minimize such pollution and should be utilized. The EPA rules and regulations for feedlot point source pollution problems should provide the necessary control of runoff from livestock operations that are intensive animal feedlots and from operations which have components which approach feedlot conditions, i.e., barnlots and manure storage areas.

A narrower, arbitrary definition of an intensive animal feedlot was used in this report to locate the large livestock operations in the Basin: cattle - 100 or more head, hogs or pigs - 200 or more capacity, and poultry -

10,000 or more birds. Available data did not permit separate identification of beef and dairy cattle operations. The location of such large operations has been noted in Section 3, Figures 1-4. The large livestock operations are in the same areas where most of the livestock are produced, i.e., southern Wisconsin, southern Michigan, western Ohio, and northern New York.

The major drainage from land containing livestock operations enters Lake Michigan from Wisconsin and southwestern Michigan and the western end of Lake Erie from Ohio. About 50% of the dairy cattle and about 56% of the hogs and pigs in the Basin are in the Lake Michigan drainage basin.

Few livestock operations are in the Lake Superior drainage basin. Because of the small amount of land in Illinois and Indiana that is tributary to Lake Michigan or Lake Erie, animal production operations in these states are not major contributors of potential pollution in the Great Lakes Basin.

The less intensive animal production operations will continue to produce most of the livestock in the Great Lakes Basin in the forseeable future. The distribution of livestock production in the Basin, as identified in Section 3 and the Appendix, will remain the same unless there are severe alterations in feed or energy supplies. Other general trends will continue such as the trends to fewer but larger livestock operations of all categories, and a continued increase in cattle, calves, hogs, and pigs in the Basin and smaller numbers of sheep and lambs.

The major pollution potential associated with agriculture in the Basin is due to dispersed livestock operations, such as those on pasture, to land disposal of animal wastes, and to general crop production. Major land use areas in the Basin can be used as a rough estimate of the runoff problem associated with livestock operations. Pasture land, which is assumed to include land for all livestock operations, is only about 5-14% of the agricultural land in the Basin. The pollution potential of runoff from livestock operations is greater than that from crop or forest land and the above percentage may not be an accurate reflection of the proportion of pollution from animal operations in the Basin. However these estimates infer that livestock operations may not be the largest contributor of potential pollution to the Great Lakes.

Because of the EPA point source feedlot regulations and because most livestock operations in the Basin are not "intensive", potential pollution from these operations will be associated with land runoff. Non-point sources of this nature are characterized by their diffuse nature and their variability with time and location. It is difficult to collect samples of non-point discharges, to characterize their pollutants, to monitor compliance with effluent regulations, or to remove pollutants from such discharges. In addition, it is often difficult to establish a direct causual relationship between many non-point source activities and their impact on water quality or to develop meaningful unit quantitative terms.

The most promising control option is to identify the activity causing pollution and to modify the activity in order to reduce or eliminate the discharge. Because of geographical variability of the non-point sources, control programs will have to provide for variations on a regional or local basis.

Pollutants in land runoff may be dissolved and carried by water or may be absorbed and transported with sediment. Hydrologic factors describing the movement of the runoff from its source to its discharge in a stream or lake must be known to predict the extent of actual pollution. The rates of soluble and particulate transport in runoff must be considered. The mechanisms of nutrient and sediment transport and deposition on land and in waterways is known only in general terms. Knowledge of these mechanisms is not adequate to relate the extent of these losses to different waste or land management situations nor is it adequate to quantify the transformations that take place as the pollutants in runoff are transported across intermittent land and in streams. Information on runoff pollutant changes during land and water transport is essential to an understanding of the real impact of agricultural land runoff on surface water quality. Such changes will be affected by soil and land characteristics, hydrologic patterns, types and extent of vegetation on the land between the runoff source and its discharge to surface waters, waste management considerations, and the rates with which the stream can assimilate the non-point contributions.

Literature exists on the quality of runoff from livestock operations and agricultural land. However, very few of these studies directly relate the type of operation or use of land to water quality. From a water resources management view, predictive methods must be developed to relate the nature and extent of pollution from non-point sources to actual water quality. A dispersed livestock

operation will contain a number of non-point sources. Predicting the actual contribution of pollution from the entire livestock operation or watershed is difficult unless a comprehensive input-output, mass balance inventory of the pollutants is available together with reaction rates during generation, transport, and deposition in the waters of concern. At present, potentially useful data are available but their usefulness is limited by uncertainties imposed by the physical description of the operation, facility, or watershed, and by a lack of adequate, corrollary information.

Data describing the characteristics of feedlot runoff and runoff from pasture and land receiving manure is accumulating. A number of current research projects have acquisition of such data as a major portion of their objectives. However few current studies are comprehensive enough to describe the runoff in terms of per unit quantitative values or to relate such values to the operational or management conditions that caused them. Additional studies permitting development of per unit quantitative values are needed.

The scarcity and variability of definitive data suggests extreme caution in attempting to extrapolate the unit quantitative values in this report to indicate the potential pollution from livestock operations and associated land use. Available values should be best thought of as initial, best estimates.

When animal wastes are disposed of on land, resultant runoff can contain a greater quantity of pollutants than will runoff from crop or forest land. Greater information is needed on animal waste land application rates that will not result in subsequent pollution problems. It appears that if the waste application rates are limited by nitrogen application rates and if the land is managed to minimize erosion, the resultant runoff will not contain a significant amount of total contaminants. The nitrogen application rates are those that avoid excess percolation or runoff of nitrogen.

At present, groundwater pollution from livestock production appears not to be serious. Groundwater monitoring programs are meager. Localized problems can result wherever good waste management approaches are not used. Continuing attention needs to be given to this potential problem.

Existing waste management technology can avoid gross pollution problems from livestock operations. Current research will provide additional information on feasible alternatives. Guidelines and Codes should be developed and

available in each state to assist producers to utilize the more appropriate technical approaches for their operations. A flexible, intelligent pollution control enforcement program is the program best suited for animal production operations. If comprehensive enforcement programs, are thought necessary for animal wastes, additional financial and personnel support will be needed at the federal and state levels.

A brief summary of the magnitude of water pollution problems due to livestock operations in the Great Lakes Basin and estimates of the needs to minimize these problems is indicated in Table 19.

9.2 Needed Activities

It continues to be difficult to draw detailed conclusions on the potential impact of animal wastes and intensive animal feedlots on the water quality of the Great Lakes. Several actions are needed to better define and minimize such impact.

a) - Institute close scrutiny on the quality of the groundwater in and adjacent to confined livestock operations, land used for waste disposal, manure storage, and pasture. Analysis of water from existing wells may be helpful but one or more detailed groundwater monitoring programs in the Great Lakes Basin are desirable. Sufficient data should be collected to relate changes in groundwater quality to livestock and waste management conditions such as livestock density, waste application rates, and to environmental conditions such as soil characteristics, precipitation, and temperature.

b) - Develop several comprehensive surface runoff monitoring programs which will be able to relate runoff characteristics to land and waste management conditions. These are desirable to quantify runoff characteristics from confined livestock operations and pasture land. Data on physical, chemical, bacteriological, and hydrologic characteristics of the runoff should be collected. Multidisciplinary involvement will be necessary.

c) - Either as part of (b) above or separately, studies should be initiated which will quantify the changes that take place in runoff contaminants as the runoff is transported across land and in streams. Available data generally

TABLE 19
IDENTIFICATION OF WATER POLLUTION PROBLEMS AND NEEDS TO MINIMIZE THESE PROBLEMS FOR LIVESTOCK OPERATIONS IN THE GREAT LAKES BASIN

PROBLEM OR NEED	SUSPECTED* 1973	KNOWN 1973	SUSPECTED*** 1980	INEVITABLE** 1980	SUSPECTED*** 2020	INEVITABLE** 2020
PROBLEM						
Runoff from						
-intensive animal feedlot						
(basin wide)	moderate		low		low	
(local)	serious		moderate		low	
-barnlots	moderate		low		low	
-land used for manure disposal	moderate		moderate		low	
-pasture	low		low		low	
Nutrient load	moderate		moderate		low	
Bacterial contamination	moderate		moderate		moderate	
Groundwater contamination	low		low		low	
Regulatory,						
-enforcement	serious		serious		moderate	
-legislation inadequate	low		low		low	
NEED						
Increased technology		moderate	moderate		?	
Unit quantitative values		serious	serious		?	
Relationships between waste management, land disposal and runoff quality		serious	serious		?	
Effective waste management guidelines		serious	moderate		?	

TABLE 19 continued

NEED	SUSPECTED* 1973	KNOWN 1973	SUSPECTED*** 1980	INEVITABLE** 1980	SUSPECTED*** 2020	INEVITABLE** 2020
Personnel to assure pollution control and to design systems		serious	serious		?	
General public education		moderate	moderate		?	

*Because of minimum information about specific problems in the Great Lakes Basin, the magnitude of the problems are not known and can only be suspected. The estimate of the suspected severity is noted.

**The term inevitable is not applicable to this estimate since it is likely that some level of a noted problem will always occur. Complete elimination of one of the problems is not likely. The estimated severity of respective problems in the subsequent years is noted.

***The severity in future years is estimated assuming application of only the regulations, guidelines, and research noted in this report.

characterizes runoff as it leaves a plot of land or a livestock operation. Changes will take place in transport with the result that different quantities of pollutants will reach the ultimate surface water receptor.

d) - In all studies but especially those in (b) and (c) above, sufficient information should be collected so that the results can be presented in quantitative units such as quantity of contaminant per unit land area or amount of runoff. Such units should be correlated to the land and waste management conditions that were a cause of the runoff and its contaminants.

e) - Research studies continue to be needed to identify the initial feasibility of new technological approaches for livestock waste management. Federal, state, and producer support should be provided for such studies.

The urgent research need is to identify land waste application rates and techniques that minimize surface and groundwater pollution problems, and on waste treatment approaches that minimize the quantity of pollutants that are disposed of on the land. Such studies should be funded at the large scale, demonstration project level. Multidisciplinary participation is desirable.

All animal waste management projects should provide information on the economic as well as the technical feasibility of the approaches.

f) - If not available, each state in the Great Lakes Basin should develop good practice guidelines to advise producers on approaches to avoid water pollution problems.

g) - No additional legislation appears necessary to control pollution from livestock operations. NPDES and SPDES activities should be adequate to identify operations in need of scrutiny and control. The EPA regulations for the feedlot point source category appear adequate to control surface water pollution problems. State enforcement activities for livestock operations need not be more stringent than the federal activities.

A need does exist to develop approaches to minimize pollution from the non-point sources at livestock operations. The research and guidelines noted above can be of significant help to meet this need.

h) - A need exists to educate the public, livestock

producers, and pollution control specialists about the magnitude of the problem, relative costs, and the availability of feasible approaches. More extensive support should be provided for such education and especially the education of professionals trained in agricultural waste management.

Section 10

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APPENDIX

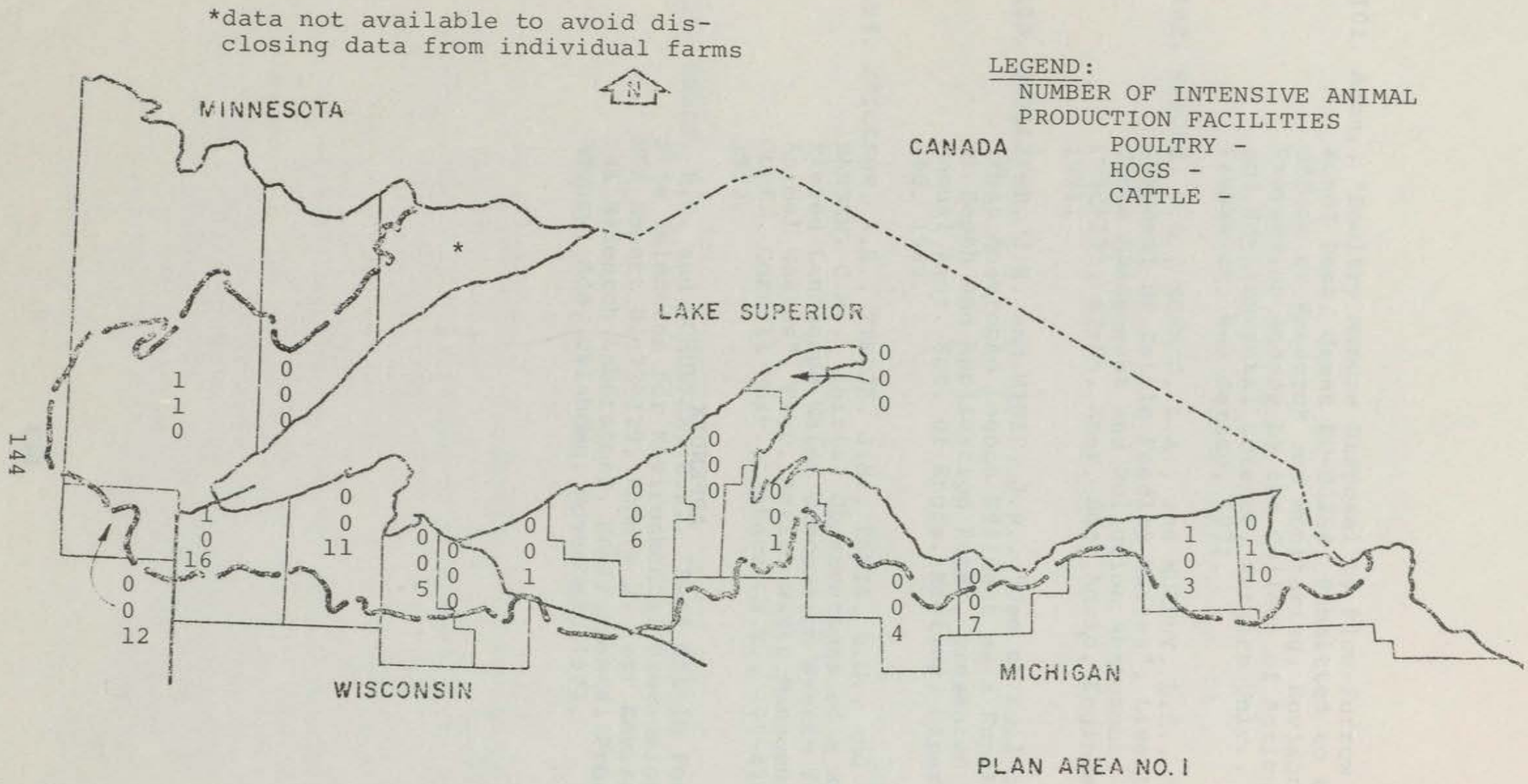
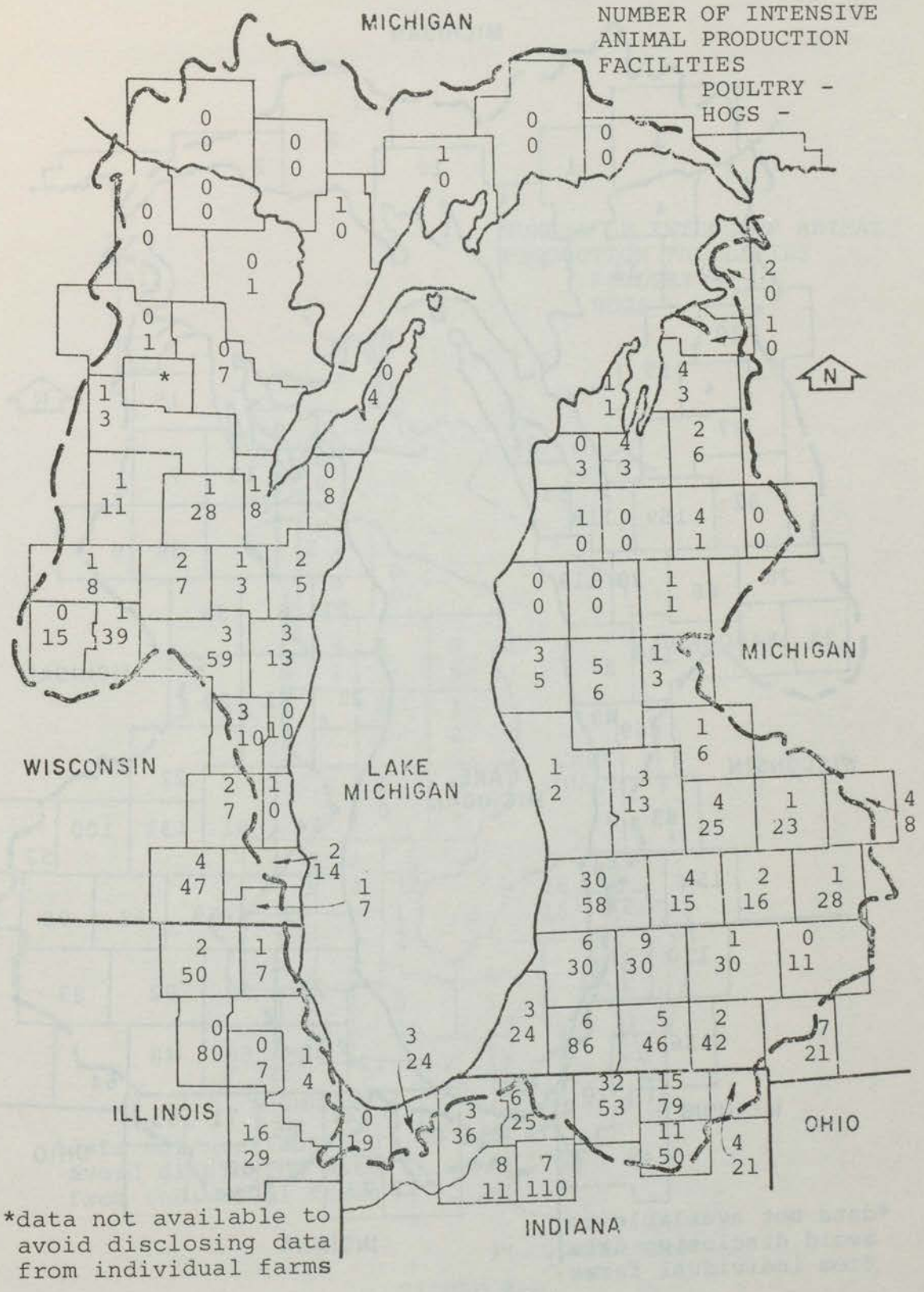
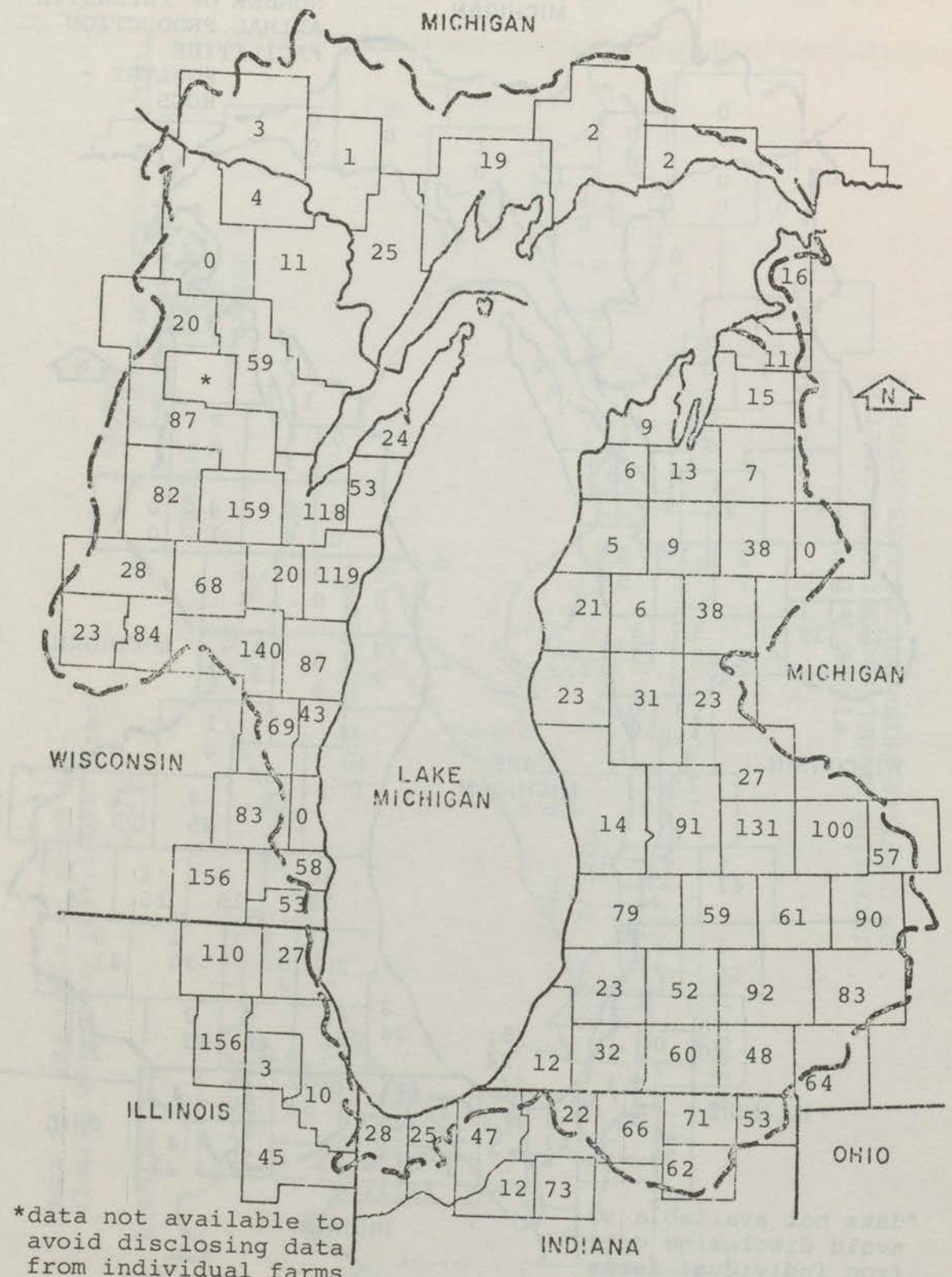


FIGURE A-1
INTENSIVE ANIMAL PRODUCTION FACILITIES
IN COUNTIES TRIBUTARY TO LAKE SUPERIOR



PLAN AREA NO. 2

FIGURE A-2
INTENSIVE POULTRY AND HOG PRODUCTION FACILITIES IN COUNTIES TRIBUTARY TO LAKE MICHIGAN



PLAN AREA NO. 2

FIGURE A-3
NUMBER OF INTENSIVE CATTLE PRODUCTION FACILITIES IN COUNTIES TRIBUTARY TO LAKE MICHIGAN

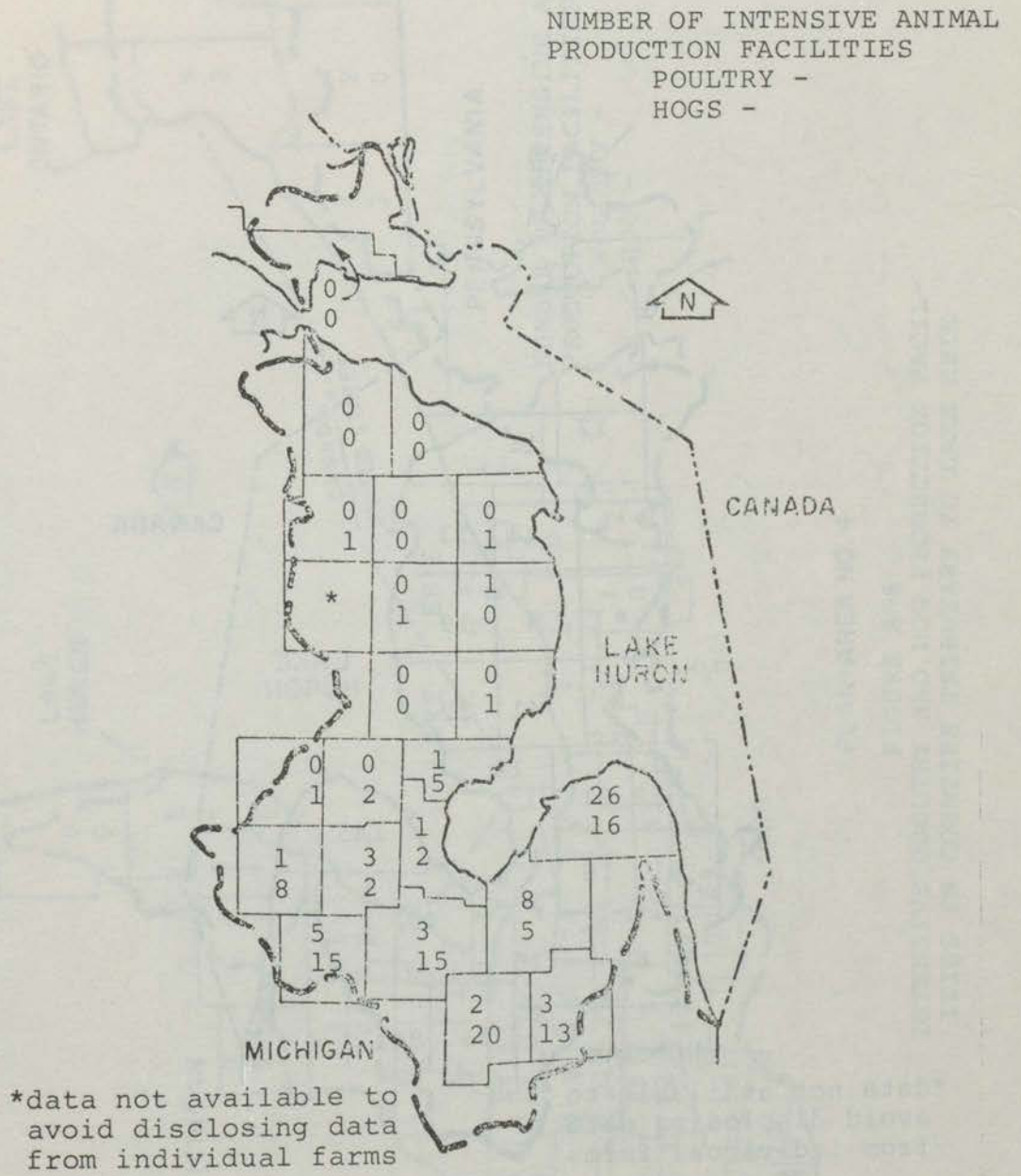
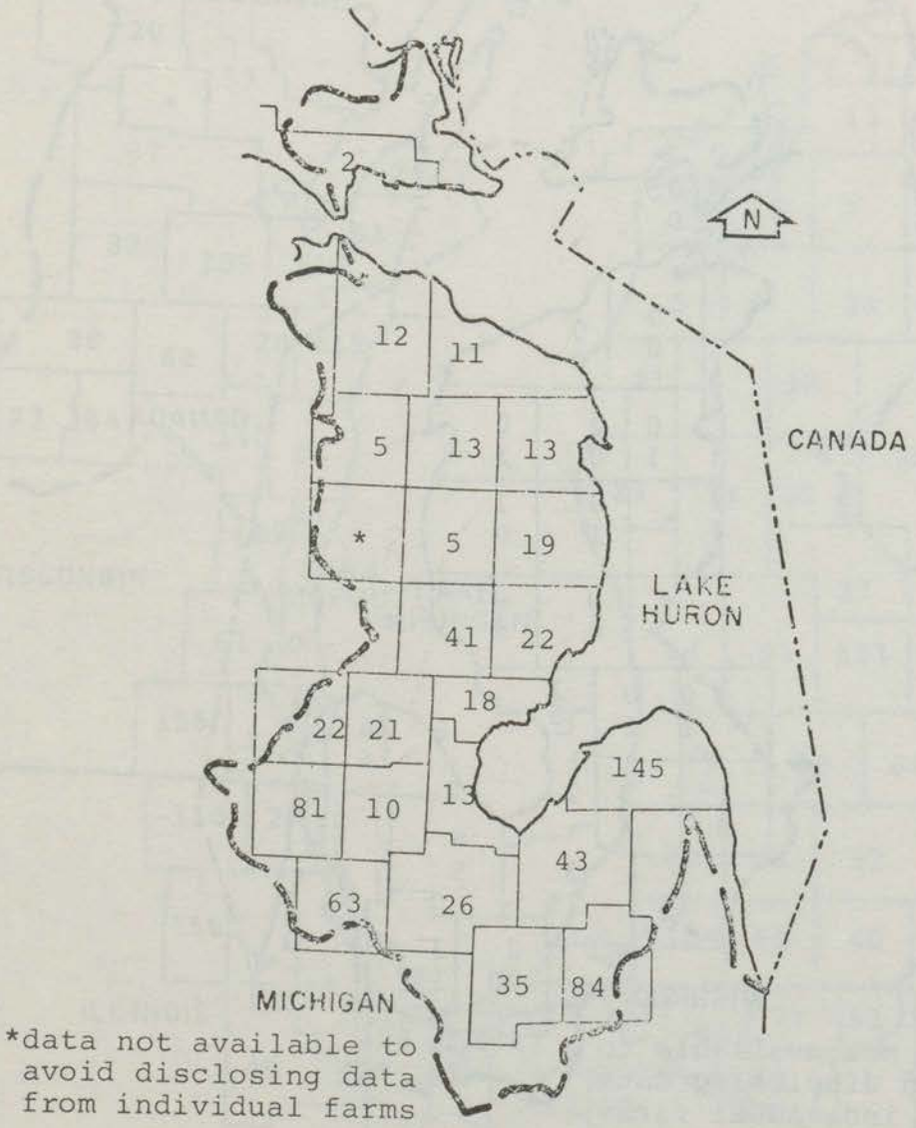
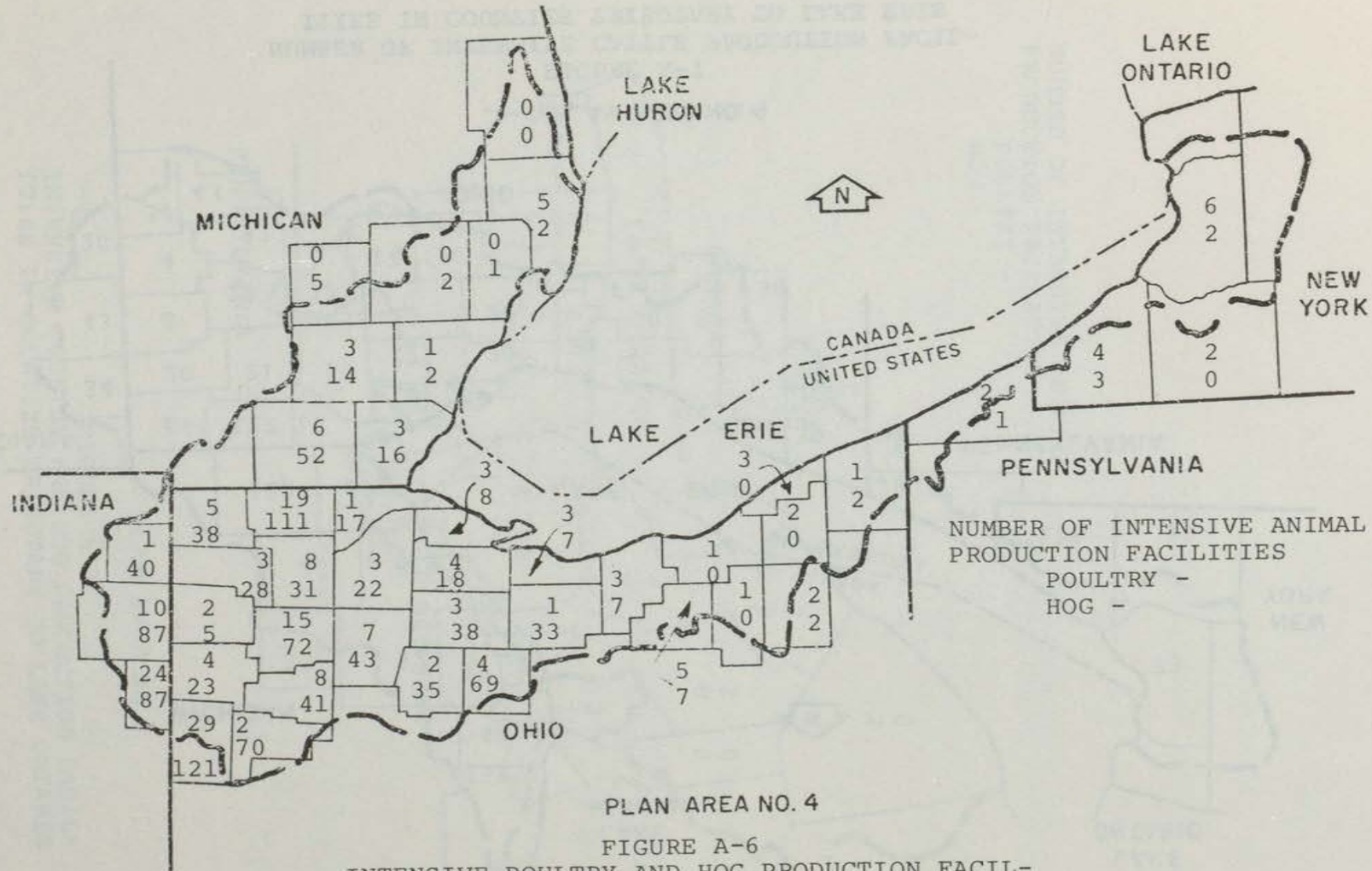


FIGURE A-4
INTENSIVE POULTRY AND HOG PRODUCTION FACILITIES IN COUNTIES TRIBUTARY TO LAKE HURON



PLAN AREA NO. 3

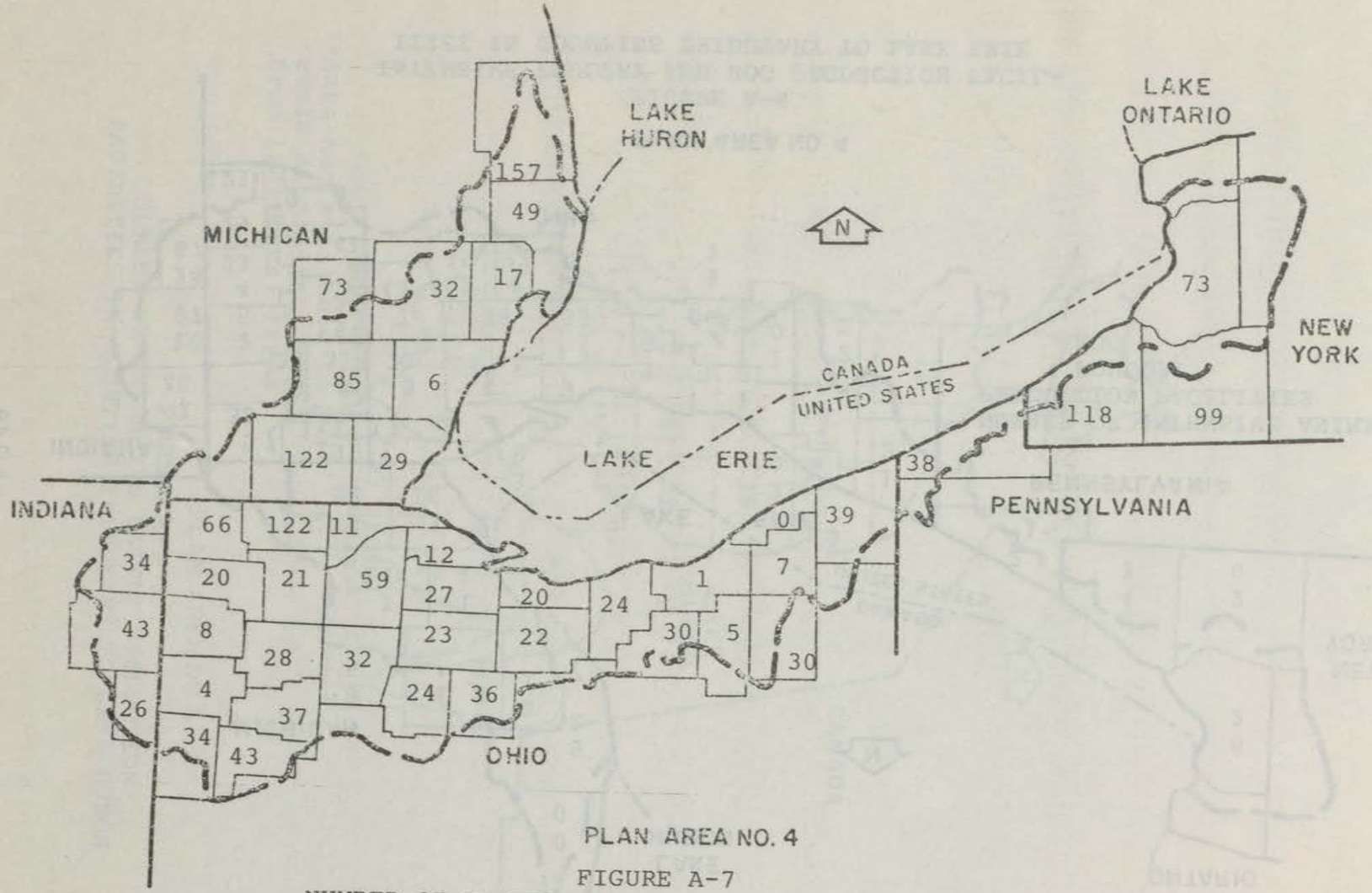
FIGURE A-5
NUMBER OF INTENSIVE CATTLE PRODUCTION
FACILITIES IN COUNTIES TRIBUTARY TO LAKE HURON



NUMBER OF INTENSIVE ANIMAL PRODUCTION FACILITIES
POULTRY -
HOG -

PLAN AREA NO. 4

FIGURE A-6
INTENSIVE POULTRY AND HOG PRODUCTION FACILITIES IN COUNTIES TRIBUTARY TO LAKE ERIE



PLAN AREA NO. 4

FIGURE A-7
NUMBER OF INTENSIVE CATTLE PRODUCTION FACILITIES IN COUNTIES TRIBUTARY TO LAKE ERIE

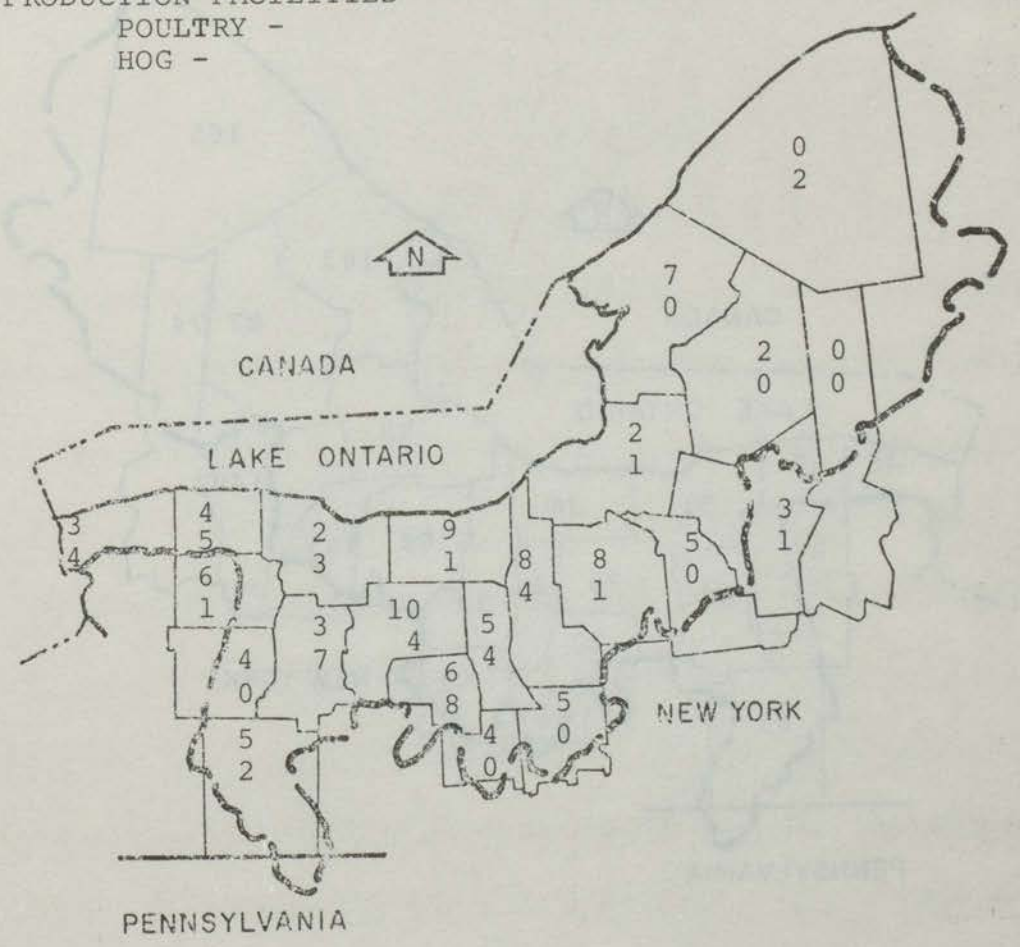
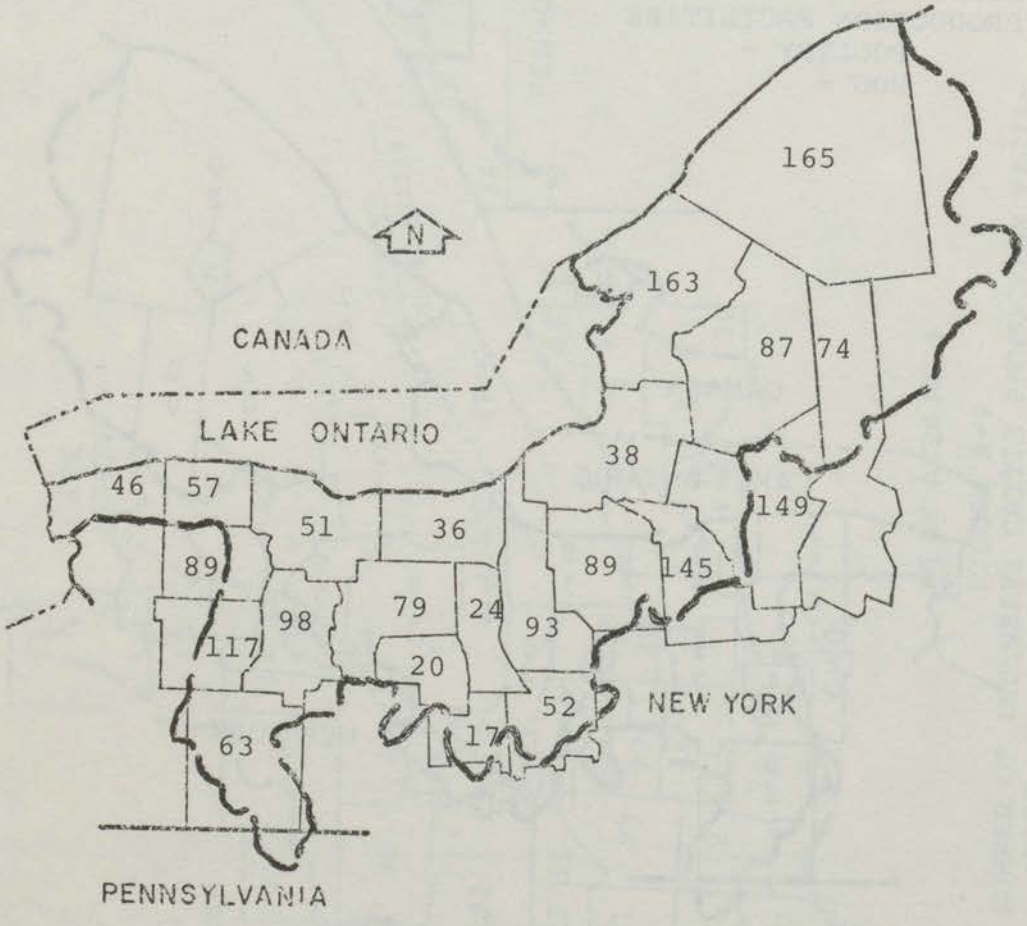


FIGURE A-8
INTENSIVE POULTRY AND HOG PRODUCTION FACILITIES IN COUNTIES TRIBUTARY TO LAKE ONTARIO



PLAN AREA NO. 5

FIGURE A-9

NUMBER OF INTENSIVE CATTLE PRODUCTION FACILITIES IN COUNTIES TRIBUTARY TO LAKE ONTARIO